Assessment of the Emissions and Energy Impacts of Biomass and Biogas Use in California

Provided to the California Air Resources Board
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Abstract

Biomass contributes more than 5,700 Gigawatt-hour to California's instate renewable power, approximately 19% of in-state renewable power and 2% of full California power mix. Current operating biopower capacity is about 900 Megawatt (MW), including approximately 550 MW of woody biomass solid fuel combustion, 280 MW of landfill gas-to-energy and 75 MW from wastewater treatment biogas. It is estimated that there is sufficient in-state 'technically' recoverable biomass to support another 2,800 MW of capacity or 21 Terawatt-hour of electricity. While most biomass energy is derived from woody material (including urban wood waste, forest product residue as well as agricultural residues), there is a growing interest in using municipal solid waste, food processing waste, increased use of animal manures and applying co-digestion techniques at wastewater treatment facilities to generate electricity and renewable fuels. Increasing production of bioenergy contributes to energy sustainability while reducing greenhouse gas emissions and could help reduce criteria pollutant emissions.

This study assesses the air quality impacts of new and existing bioenergy capacity throughout the state, focusing on feedstocks, and advanced technologies utilizing biomass resources predominant in each region. The options for bioresources include the production of biopower, renewable NG and ethanol. Emissions of criteria pollutants and greenhouse gases are evaluated for a set of scenarios that span the emission factors for power generation, and the uses of renewable natural gas for vehicle fueling and pipeline injection. Emission factors combined with the geospatially-resolved bioenergy outputs (facility locations) are used to generate new emission source locations and magnitudes which are input to the Community Multiscale Air Quality model (CMAQ) to predict regional and statewide temporal air quality impacts from the biopower scenarios.

With current technology and at the emission levels of current installations, maximum biopower production could increase NO_X emissions by 10% in 2020, which would cause increases in ozone and PM concentrations in large areas of the Central Valley where ozone and PM concentrations exceed air quality standards constantly throughout the year. Negative effects on PM would be expected in both summer and winter episodes. Among the alternatives for biomass use, technology upgrades would significantly reduce criteria pollutant emissions. Conversion of biomass to CNG for vehicles would achieve comparable emission reductions of criteria pollutants and minimize emissions of greenhouse gases. As suggested by the analysis of emissions, applying technological changes and emission controls would minimize the air quality impacts of biopower generation. And a shift from biopower production to CNG production for vehicles would reduce emissions and air quality impacts further. From a co-benefits standpoint, CNG production for vehicles appears to provide the benefits in terms of air pollutant and GHG emissions, and air quality.

This investigation provides a consistent analysis of air quality impacts and greenhouse gases emissions for scenarios examining increased biomass use in California. The findings will help inform policy makers and industry with respect to further development and direction of biomass policy and bioenergy technology alternatives needed to meet energy and environmental goals in California.

Acronyms

Acronym	Definition
AB	Assembly Bill
BACT	Best Available Control Technology
BDT	Bone-dry ton
BEV	Battery Electric Vehicle
BFB	bubbling fluidized bed
BIGCC	Biomass Integrated Gasification Combined Cycle
Btu	British Thermal Unit
CARB	California Air Resources Board
CARFG	California Reformulated Gasoline
CBC	California Biomass Collaborative
CCS	Carbon Capture and Sequestration
CFB	circulating fluidized bed
CH4	Methane
CHP	Combined Heating and Power
CMAQ	Community Multiscale Air Quality model
CO	carbon monoxide
CO2	carbon dioxide
CPUC	California Public Utilities Commission
CV	conventional vehicles
DOE	Department of Energy
DOT	Department of Transportation
EISA	Energy Independence and Security Act
EMFAC	Emission Factor model
EPRI	Electric Power Research Institute
FFV	Flex-fuel Vehicle
g/kWH	grams per kilowatt-hour
GHG	greenhouse gases
H2	molecular hydrogen
HFCV	hydrogen fuel cell vehicle
HSAD	high-solid anaerobic digestion
IGCC	integrated gasification combined cycle
IOU	investor-owned utilities
LCFS	low carbon fuel standard
LDV	light-duty vehicle
LFG	landfill gas
LUCs	land-use changes
MJ/Nm3	megajoule per normal cubic meter
MMBtu	million British thermal units
MMT	million tons
MSW	municipal solid waste

MW	megawatts
MWth	megawatts of thermal output
NG	natural gas
NGCC	natural gas combined cycle
NMHC	non-methane hydrocarbons
NOx	nitrogen oxides
NRC	National Research Council
NREL	National Renewable Energy Laboratory
OFMSW	organic fraction of municipal solid waste
PM	particulate matter
RFS	renewable fuel standards
RPS	renewable portfolio standards
RSNG	renewable synthetic natural gas
SB	Senate Bill
SoCAB	South Coast Air Basin of California
SOx	oxides of sulfur
U.S. EPA	United States Environmental Protection Agency
USDA	United States Department of Agriculture
WWTP	Wastewater treatment plant

Executive Summary

This study assesses the air quality impacts of new and existing bioenergy capacity throughout the state, focusing on feedstocks, and advanced technologies utilizing biomass resources predominant in each region. The options for bioresources include the production of biopower, renewable NG and ethanol. Emissions of criteria pollutants and greenhouse gases are evaluated for a set of scenarios that span the emission factors for power generation, and the uses of renewable natural gas for vehicle fueling and pipeline injection. Emission factors combined with the geospatially-resolved bioenergy outputs (facility locations) are used to generate new emission source locations and magnitudes which are input to the Community Multiscale Air Quality model (CMAQ) to predict regional and statewide temporal air quality impacts from the biopower scenarios.

The list of scenarios evaluated in this study explores the potential impacts of widespread implementation of biopower driven by regulatory measures and initiatives in place in California: SB1122 requires the CPUC to direct electrical corporations (IOUs) to procure 250 MW (cumulative, state wide) of new small biopower (less than 3 MW per project) in a separate IOU feed-in tariff program, of which 110 MW is for urban biogas and 90 MW for dairy and other agricultural bioenergy (that would include digester gas or small thermochemical conversion). Governor Brown's Clean Energy Jobs Plan calls for 20 GW of new renewable generation by 2020. All these measures provide a pathway to use bioresources in the state within the maximum potential. Figure ES1 provides a summary of potential installed capacity of biopower under different scenarios. Maximum potential for biopower is nearly 4,800 MW.

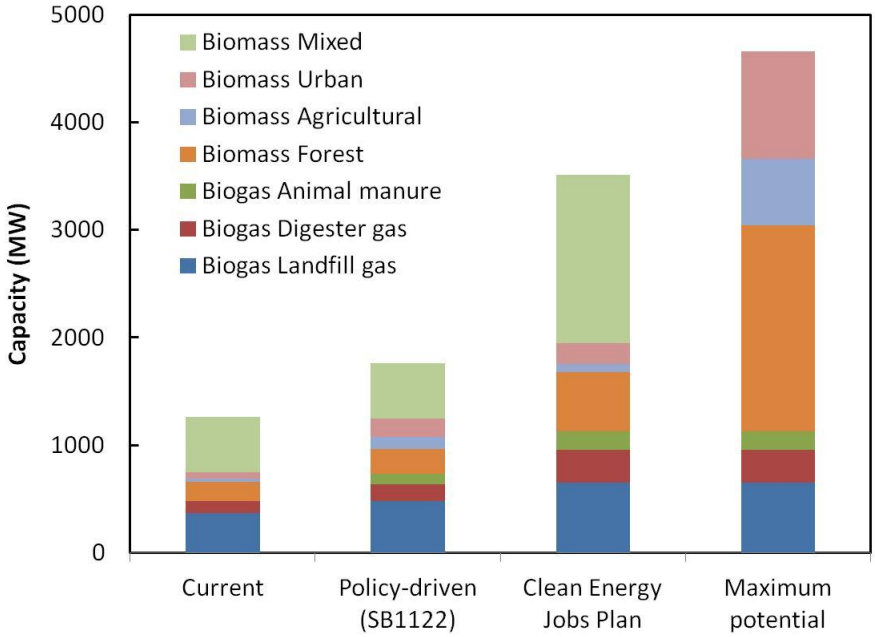


Figure ES1: Summary of power generation capacity from biomass in scenarios with current biomass technology

An alternative use of bioresources is to produce biomethane that can fuel vehicles, and contribute to the production of renewable fuels. Biomethane can be obtained via clean-up of landfill gas and anaerobic digestion biogas. In addition, biomethane can be obtained via gasification of solid biomass and production of renewable synthetic natural gas.

Table ES1 presents the maximum potential for biomethane production via RSNG from biogas and biomass resources in the state of California, and potential for cellulosic ethanol and biomethane from HSAD from solid residue. The total biomethane potential from biogas and biomass is more than $1.1 \cdot 10^6$ MMBtu/day. This amount could potentially meet fuel demand of nearly 16% of gasoline vehicles in California. Conversely, taking into account that CA reformulated gasoline (CARFG) is a blend of 5.7% ethanol and gasoline, bioethanol production from solid biomass could meet the entire state demand for ethanol blending for CARFG.

Table ES1. Maximum potential for biomethane production from biogas and biomass, and potential for cellulosic ethanol production from solid biomass

		Biogas Potential (MMBtu/day)
Biogas	Landfill gas	177424
	Digester gas	83253
	Animal manure	47768
	Total	308445

		Biomass Potential (BDT/day)	RSNG Potential (MMBtu/day)	Ethanol Potential (gal/day)	HSAD CNG (MMBtu/day)
Biomass	Forest	30668	461110	2499430	
	Agricultural	10989	165231	382069	12414
	Urban	20679	213445	475769	11354
	Total	62336	839785	3357269	
Total			1148230		23768

Emissions of criteria pollutants and greenhouse gases are evaluated for all scenarios in order to evaluate the co-benefits of using biomass for both air quality and climate change. Figure ES2 presents the emissions from a case with Technology Upgrade for Efficiency and Emissions, in comparison with the case with maximum potential for biopower with current technology. Technology upgrades consist of switching current boilers and engines with next generation gasification systems and fuel cells. The result is a significant decrease in direct emissions of criteria pollutants with respect to the case with current technology. Direct GHG emissions do not change, as the same amount of carbon is converted into CO_2, but because of the increase in efficiency in power generation, emission savings are also increased with respect to the case with maximum potential and current technology.

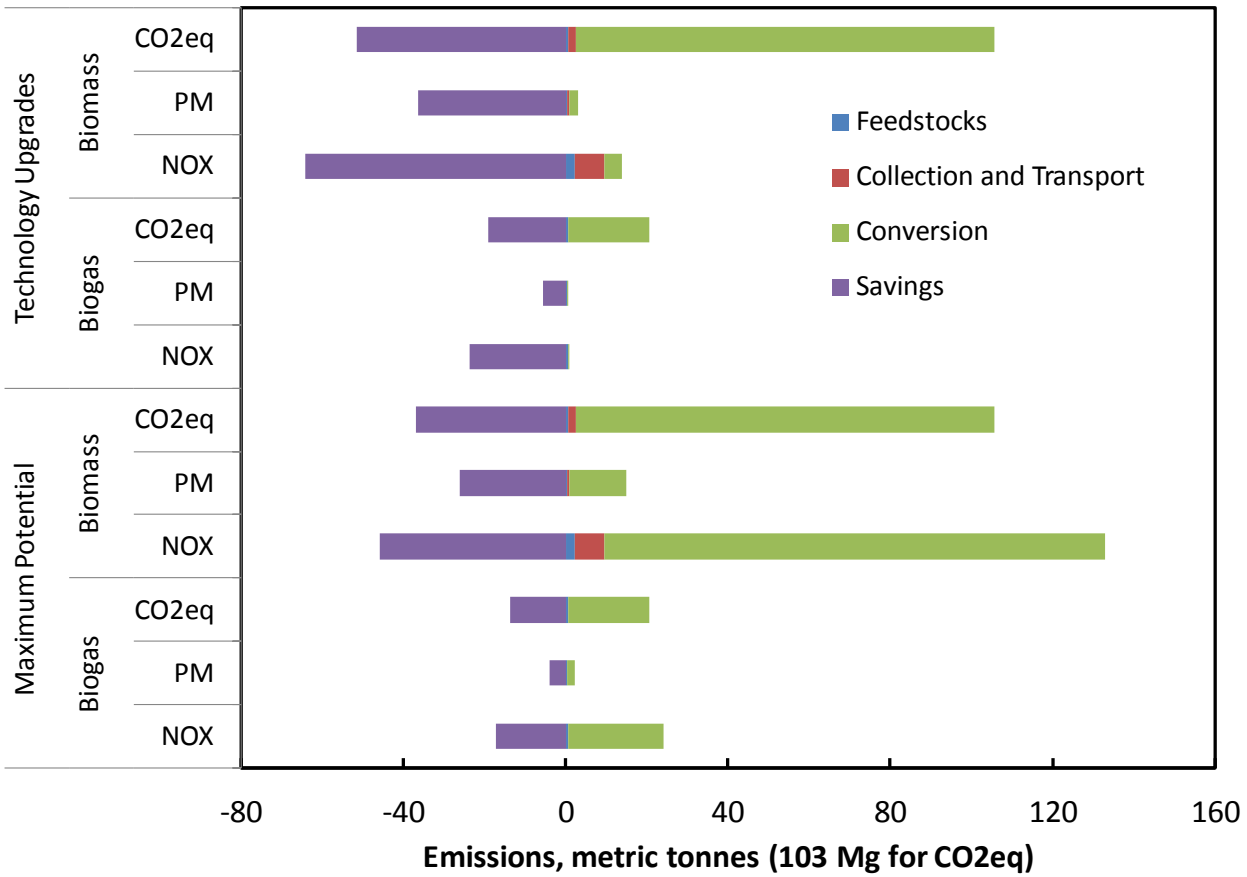


Figure ES2: Comparison of emissions from biomass in scenarios with maximum biomass potential with current technology and with technology upgrades for efficiency and emissions

Figure ES3 presents the emissions of scenarios the present a shift in the end use of biomass from electricity to fuel, together with the case with maximum potential for biopower with current technology. Group C includes two cases with generation of CNG from solid biomass via gasification: one dedicated to produce CNG for vehicle consumption and the other one for pipeline injection. Direct emissions from these two cases are the same, because the processes to generate the CNG are the same in both cases. Emissions from feedstocks in these two cases are considerably higher than in the cases of group A and B, because more energy is required to clean-up biogas and synthesis gas, and to compress them. The only difference between these two CNG scenarios is the emissions displaced by the CNG. In the case that CNG is dedicated to vehicle consumption, emission displacement is due to the savings in gasoline production and marketing needs that production of CNG from biomass provides. In addition, the case includes savings in emissions from vehicles switching from gasoline to CNG consumption. Conversely, in the case that CNG is dedicated to pipeline injection, emission displacement is calculated from the savings in natural gas production and marketing demand that CNG provides. No additional savings are considered in this CNG case as combustion of NG from biomass is assumed to

produce the same pollutant emissions as combustion of conventional NG. Hence, comparing the two cases is analogous to contrasting emissions from equivalent energy units of gasoline and natural gas. The result is that producing gasoline for California is more pollutant-intensive than producing natural gas, and thus, reducing gasoline production achieves higher emission savings than reducing production of natural gas containing the same amount of energy. Consequently, on a full fuel cycle emissions standpoint, producing CNG for vehicles is more favorable than producing natural gas for pipeline injection as shown in Figure 28.

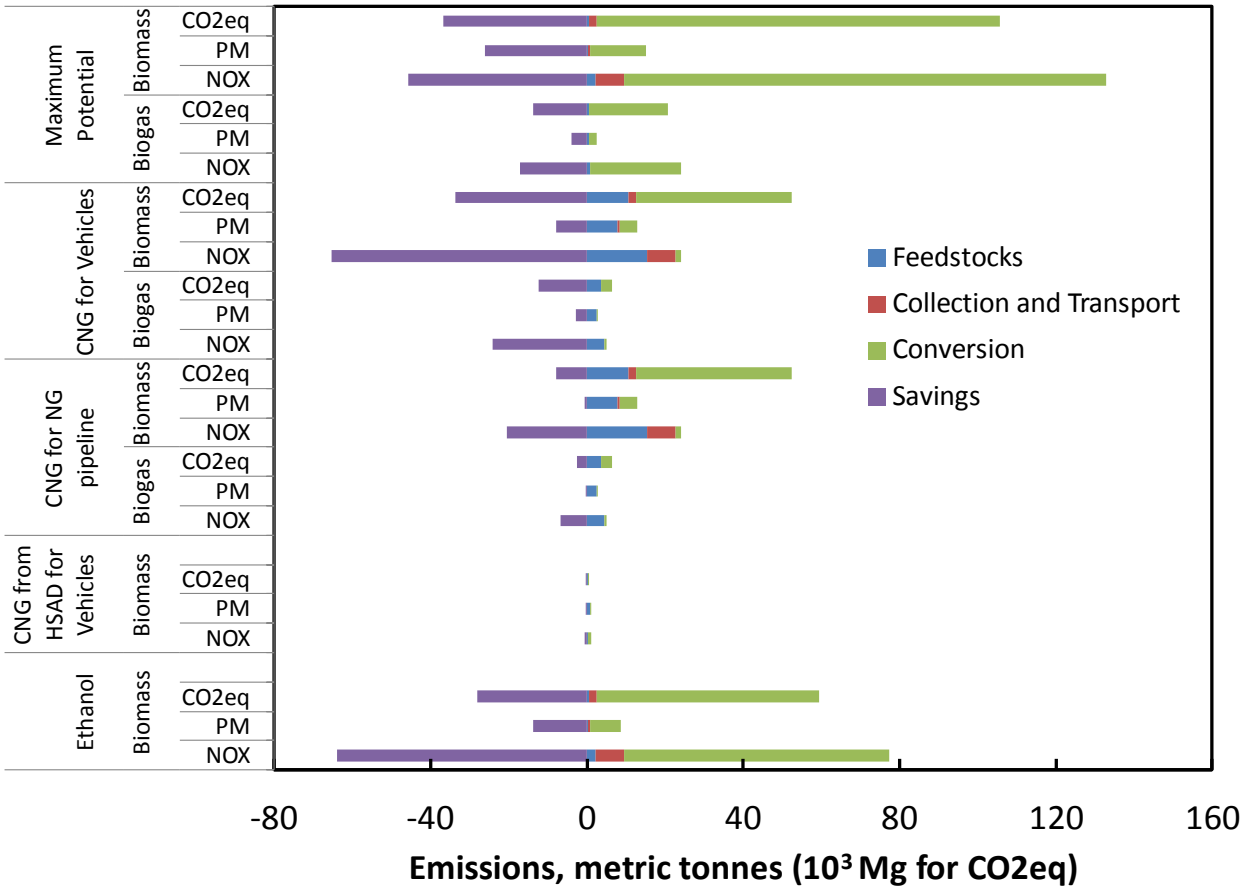


Figure ES3: Comparison of emissions from biomass in scenarios with maximum biomass potential using current technology for biopower (group A) and scenarios with CNG production (group C)

Table ES2 presents the total emissions of scenarios that assume maximum potential for biomass use. In summary, from a full fuel cycle perspective, use of biomass to produce vehicle fuels appears as the best option to minimize GHG emissions. Applying technology upgrades and emission controls for biopower production can mitigate criteria pollutant emissions, but CNG from biogas and gasification of biomass achieves comparable emissions of criteria pollutants and lower GHG emissions. An important aspect to note about the full cycle analysis is that a large portion of emission savings for criteria pollutants occur outside the state. If only the emission savings within the state are accounted for (Table ES3), the case with technological advances for

biopower production becomes the most favorable scenario to minimize the impact of biomass use on criteria pollutant emissions but CNG scenarios are still the most favorable for greenhouse gases emissions. Air quality modeling of the emission impacts in the state completes the analysis for the overall air quality impacts of biomass use.

Table ES2: Summary of net emissions from selected scenarios (in tons/day for NO_X and PM, and 10^3 tons/day for $CO_{2,eq}$)

		Maximum Potential	Technology Upgrades	Ethanol	CNG from HSAD for Vehicles	CNG for Pipeline Injection	CNG for Vehicles
Biogas	**NO_X**	6.9	-22.7			-1.8	-19.1
	PM	-1.8	-5.2			2.6	-0.1
	CO2eq	7.0	1.7			3.7	-6.2
Biomass	**NO_X**	87.2	-50.1	13.6	0.4	3.4	-41.6
	PM	-11.0	-33.3	-5.2	0.7	12.3	5.0
	CO2eq	68.9	54.1	31.1	0.1	44.6	18.5

Table ES3: Summary of net emissions from selected scenarios (in tons/day for NO_X and PM, and 10^3 tons/day for $CO_{2,eq}$) accounting only for in-state savings

		Maximum Potential	Technology Upgrades	Ethanol	CNG from HSAD for Vehicles	CNG for Pipeline Injection	CNG for Vehicles
Biogas	**NO_X**	16.0	-10.1			4.0	-1.0
	PM	0.5	-2.1			2.7	1.7
	CO2eq	12.0	8.7			5.9	-3.0
Biomass	**NO_X**	111.6	-16.0	77.5	0.9	20.9	7.7
	PM	3.6	-12.8	8.6	0.7	12.8	10.0
	CO2eq	82.4	73.0	59.3	0.3	51.2	27.2

The emissions resulting from the biomass facilities are spatially allocated in the modeling domain. For the air quality impacts it is assumed that the existing facilities will absorb the increase in biomass capacity. The increase in biopower capacity assumed in the maximum potential biopower cases is then scaled up from the existing facilities. In addition to emissions from conversion, emissions from forest residue collection are also included. The spatial allocation of collection and transport is based on the forest residue potential at a county level and

location of rural and urban roads in each county. Figure ES4 illustrate the spatial allocation of biopower facilities and collection and transport of forest residue.

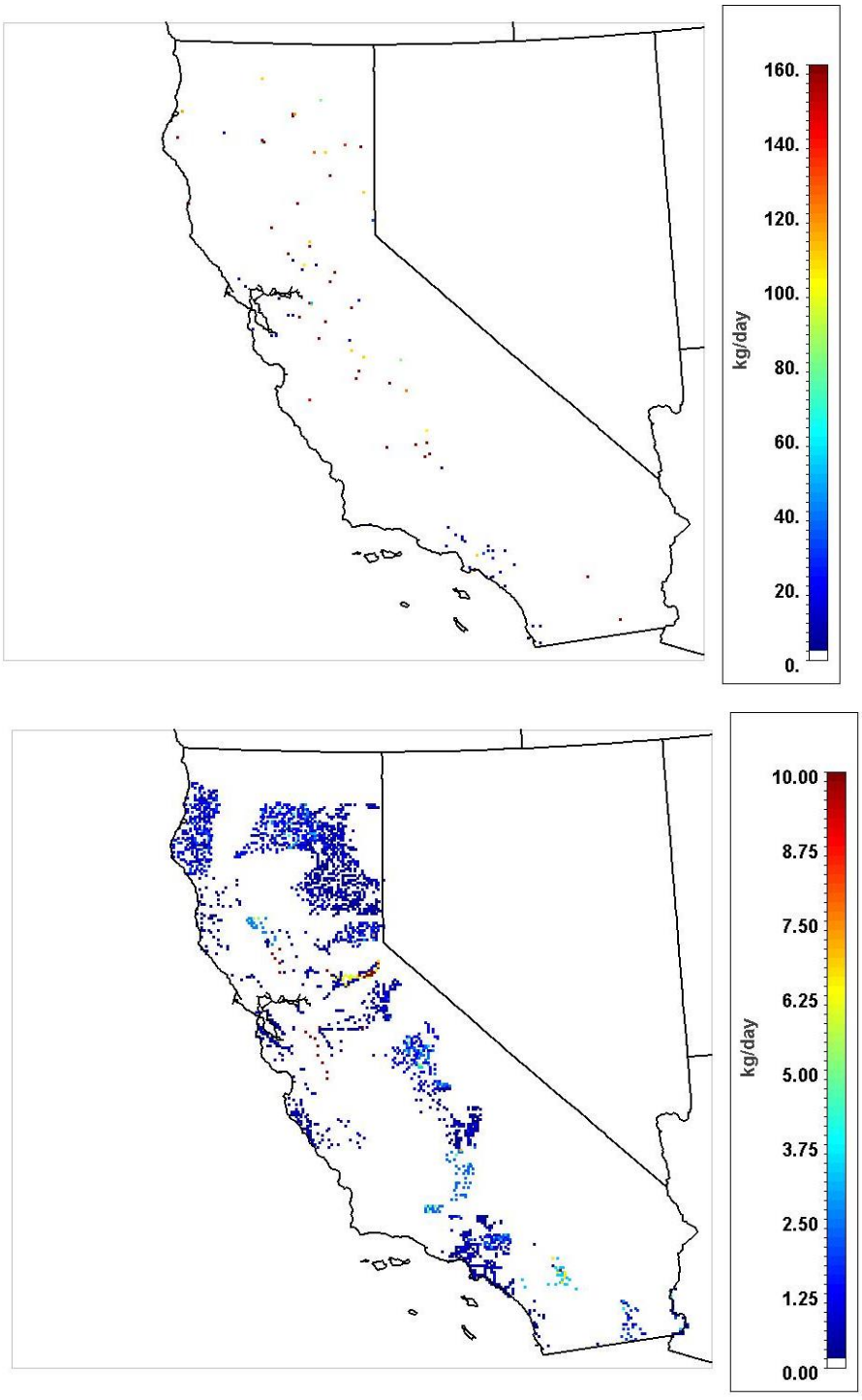


Figure ES4: Locations of emissions from biopower production for the Maximum potential for biopower production with current technology (group A). Top: NO_X emissions from biopower facilities. Bottom: NO_X emissions from forest residue collection

From the technically recoverable biomass resources, there is a potential for up to 4.66 GW of biopower that could be installed in the state. With current technology and at the emission levels of current installations, maximum biopower production could increase NO_X emissions by 10% in 2020. Among the alternatives for biomass use, technology upgrades would significantly reduce criteria pollutant emissions. Conversion of biomass to CNG for vehicles would achieve comparable emission reductions of criteria pollutants and minimize emissions of greenhouse gases.

Emission factors combined with the geospatially-resolved bioenergy outputs (facility locations) are used to generate new emission source locations and magnitudes which are input to the Community Multiscale Air Quality model (CMAQ) to predict regional and statewide temporal air quality impacts from the biopower scenarios. Installing the maximum potential of biopower production with current technology by 2020 would cause increases of over 6 ppb in ozone (shown in Figure ES5) and 2 $\mu g/m^3$ in PM concentrations in large areas of the Central Valley where ozone and PM concentrations exceed air quality standards constantly throughout the year. Negative effects on PM would be expected in both summer and winter episodes. As suggested by the analysis of emissions, applying technological changes and emission controls would minimize the air quality impacts of biopower generation. And a shift from biopower production to CNG production for vehicles would reduce emissions and air quality impacts further. From a co-benefits standpoint, CNG production for vehicles appears to provide the benefits in terms of air pollutant and GHG emissions, and air quality.

It is clear that the state has enough bioresources to meet the goals of SB1122 and Governor's plan for renewable power, and that biomass could be a large contributor to the renewable portfolio standard for the state. However, if California is to meet the air quality goals for non-attainment areas like the San Joaquin Valley, it should minimize the impact of using biomass with advanced technologies like fuel cells for biogas and gasification systems for solid residue.

This investigation provides a consistent analysis of air quality impacts and greenhouse gases emissions for scenarios examining increased biomass use. The findings will help inform policy makers and industry with respect to further development and direction of biomass policy and bioenergy technology alternatives needed to meet energy and environmental goals in California.

Future research needs should include the collection of more specific emission factors and better characterization of processes for advanced technologies, such as production of renewable synthetic natural gas. For the analysis presented here, emissions and energy balances from generic gasification facilities were assumed. Another area of research related to biomass use would be the in-depth analysis of management of solid waste to maximize recycling, and minimize disposal at landfills. These management strategies could require additional infrastructure and reduce the biogas and biopower yields from landfills.

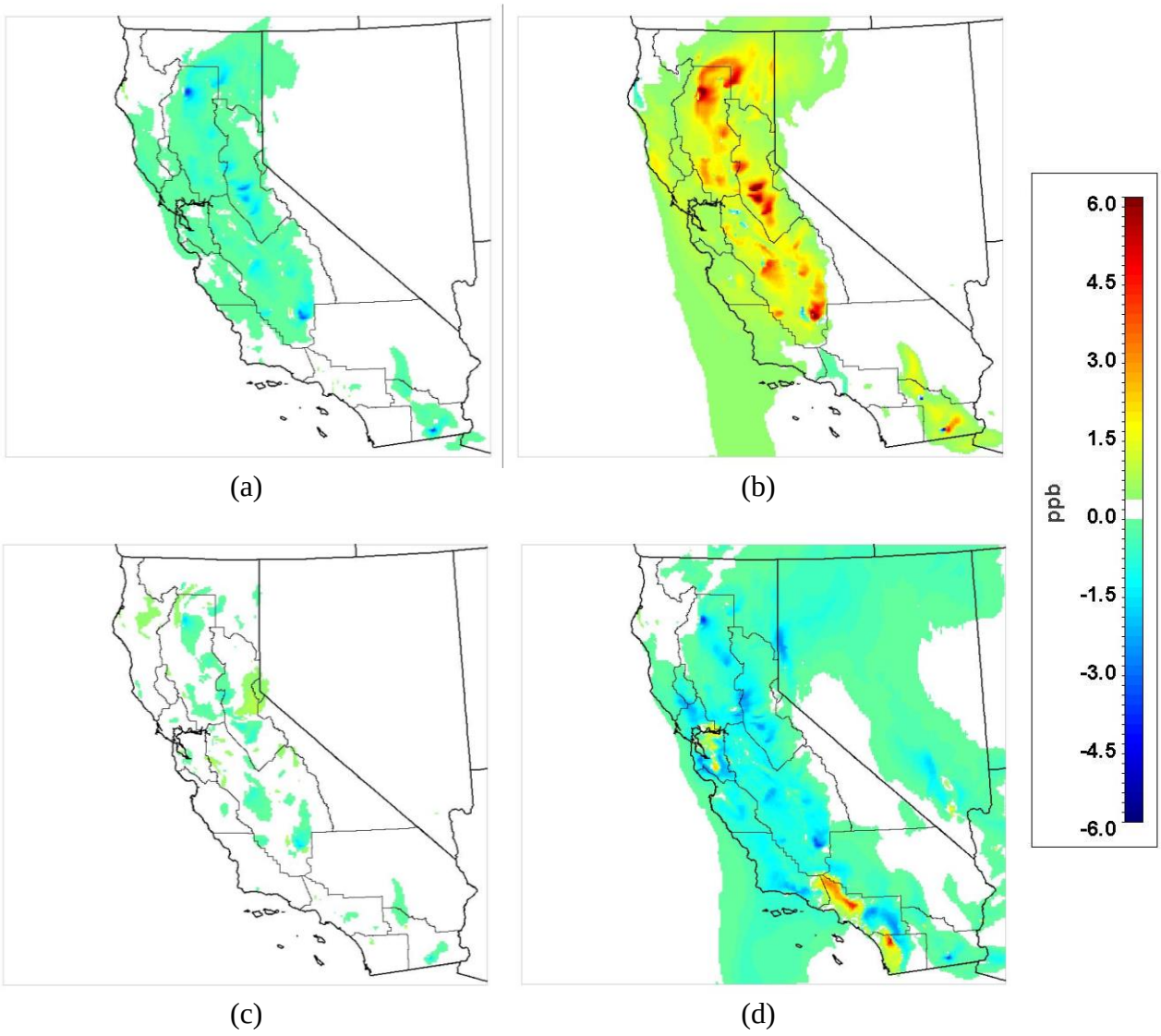


Figure ES5: Changes in peak ozone concentrations due to biomass scenarios in a summer episode with respect to the baseline case: (a) No Biomass Case, (b) Maximum biopower production with current technology, (c) Maximum biopower production with enhanced technology, (d) Maximum production of CNG from biomass for vehicle consumption.

1 Introduction

There is a growing interest in the State of California to support a clean energy future to meet the mandate of the Global Warming Solutions Act – Assembly Bill 32. California has a long history of environmental innovations and regulations that have significantly improved air quality throughout the last four decades, and there is a renewed commitment to environmental stewardship that includes reducing greenhouse gases emissions. Meeting stricter clean air standards while reducing greenhouse gas emissions will require well integrated energy and air quality programs. Renewable energy will be one of the key technologies to reduce both criteria pollutant and greenhouse gas emissions, and sustainable bioenergy can contribute to the mix of renewable energy technologies. Bioenergy technologies and resources can provide a range of economic and environmental benefits to the state. Bioenergy can be garnered from digester gas, landfill gas and biomass resources to produce electric power, heat, and/or renewable gaseous or liquid fuels. Renewable liquid or gaseous biofuels can be used for stationary or vehicular applications. The California Air Resources Board has adopted regulations to promote renewable electric power and renewable transportation fuels through the Renewable Electricity Standard and the Low-Carbon Fuel Standard. These standards require significant reductions in greenhouse gases emissions, which will require a suite of solutions that will include biomass and biogas use, among other types of renewable resources.

This modeling study assesses the potential implementation of biomass infrastructure to determine preferred uses and strategies for use of California's renewable resources. The analysis quantifies the emissions of greenhouse gases and criteria pollutants of different fuel paths for biomass and biogas management and utilization and the potential to exploit emerging biomass and biogas resources. The resulting emissions are spatially and temporally resolved for subsequent use in air quality modeling to account for atmospheric chemistry and transport to determine the overall air quality impacts of the new biomass and biogas infrastructure. The analysis of both greenhouse gases and criteria pollutants provides a scientific basis to evaluate the potential co-benefits of biomass and biogas use for air pollution control and climate change mitigation strategies.

2 Biomass Resources

Biomass contributes more than 5,700 GWh to California's instate renewable power (this is about 19% of in-state renewable power and 2% of full California power mix) (CEC 2010). Current operating biopower capacity is about 900 MW (including approximately 550 MW of woody biomass solid fuel combustion, 280 MW of landfill gas-to-energy and 75 MW from wastewater treatment biogas) (CBC, 2011). It is estimated that there is sufficient in-state 'technically'[1] recoverable biomass to support another 2,800 MW of capacity or 21 TWh of electricity (Williams et al., 2008). While most biomass energy is derived from woody material (including urban wood waste, forest product residue as well as agricultural residues), there is a growing

[1] Technical biomass resource is that which can be sustainably recovered with minimal impacts to erosion, riparian zones, soil organic matter and other agronomic factors. There is no economic filter applied to the technical resource estimate.

interest in using municipal solid waste and applying co-digestion techniques at wastewater treatment facilities to generate electricity and renewable fuels.

While much of the landfill gas (LFG) in California is collected and utilized or flared and all wastewater treatment biogas is utilized or flared, fugitive emissions (and some LFG venting) contributes to nearly 2% of the total greenhouse gases emissions in California and the U.S. Utilizing more of the currently flared biogas in the state, as well as switching or improving some of the existing biogas energy facilities can reduce criteria and greenhouse gas emissions while increasing renewable power or fuels. Utilizing waste materials as feedstocks for engineered anaerobic digesters (such as food and green waste from the MSW stream and food processor wastes) could potentially support 300 MW of electricity or 30 PJ of fuel (CBC 2011b).

Biogas can be utilized as a substitute for natural gas (after appropriate cleaning and treatment) contributing to energy sustainability while reducing greenhouse gas emissions. In addition, biogas use could help reduce criteria pollutant emissions. Upgraded biogas can be used directly in compressed natural gas vehicles or in stationary fuel cells to produce electricity and hydrogen, which can then be used as a transportation fuel for electric and hydrogen fuel cell vehicles. These vehicle technologies could reduce criteria pollutant emissions compared to combustion-based vehicles using gasoline or compressed natural gas. Methane, hydrogen and/or electricity produced from biogas will contribute to the suite of low-carbon fuels that will be necessary to meet the Low Carbon Fuel Standard (LCFS) goals.

The potential air quality impacts from the use bioresources depends on the location of those resources, how those resources are processed, the products obtained from bioresource utilization, and the technologies used in the processing of biomass. For example, forest residue can be combusted to produce power or digested to produce bioethanol for fuel. The production of biopower or biofuels from the same bioresources may result in very distinct air pollutant emissions. Similarly, biogas from landfills can be combusted in an engine to produce biopower, or it can run fuel cells without any combustion involved resulting in much lower emissions. Section 3 describes the options for biomass use. In California, most of the existing biomass facilities use bioresources to produce power, but there are already two landfill installations that generate up to 18,000 gallons per day of liquefied natural gas that fuel refuse trucks. Some other biogas installations also pipe the biogas to be used for heat production for process heating. Finally, there are 17 installations in the state that produce ethanol and biodiesel from a variety of waste streams, including corn and sorghum residue, and used oils.

Forestry, agricultural waste and urban green clippings, which constitutes the largest portion of solid biomass available in the state of California, is mostly distributed along the Central Valley and the Northern part of the state. Figure 1 presents the technical biopower potential from forestry, agricultural and urban green waste by county for 2020, and the location and capacity of the existing facilities processing that type of biomass. In the San Joaquin Valley, there is a high concentration of agricultural activities that generate high volumes of waste. The northern counties of California are populated with forests that provide a source of forestry waste that can be utilized for biopower. Table 1 presents the technology distribution of the biopower installations processing solid biomass. Approximately 49% of the biopower capacity is produced with stokers, which is the oldest technology, whereas other 45% is produced by

fluidized bed technology. One installation uses a suspension boiler for rice hulls to produce 25 MW, and another installation uses walnut shells in a gasifier to produce 100 kW. The total power produced by these installations is 638 MW, from which 155 MW are co-produced with heat for process heating. Based on estimates by the California Biomass Collaborative (Williams et al., 2008), the technical potential for biopower from solid biomass for the year 2020 is 3650 MW, more than 3000 MW additional capacity with respect to the existing capacity. The increase in potential biopower capacity assumes a significant improvement in efficiency from biomass installations from 20% to 30%.

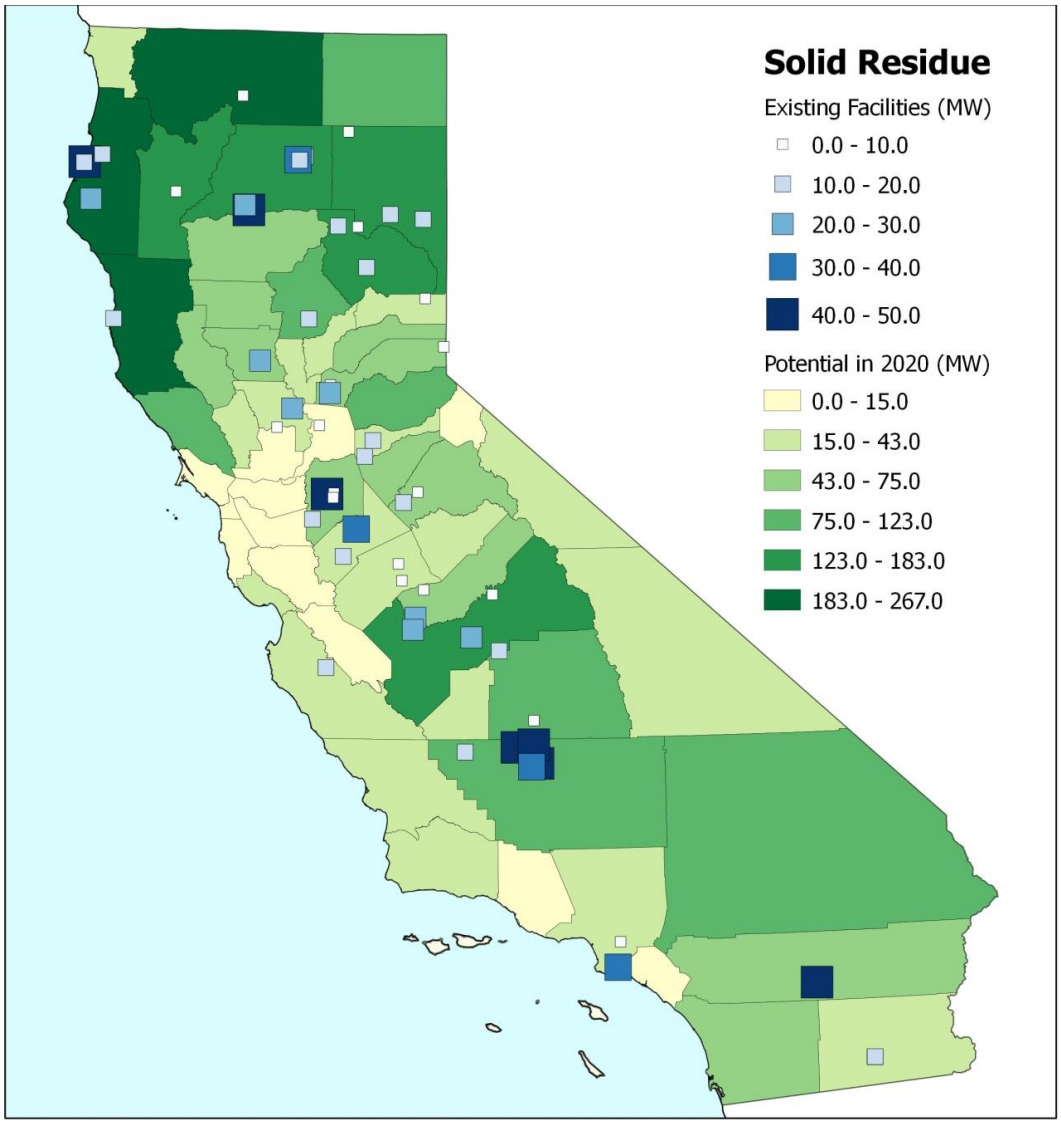


Figure 1: Solid residue potential for biopower production in 2020 and capacity and location of existing facilities in California.

Table 1: Technology distribution for biomass solid residue biopower installations

Technology	Net Capacity (MW)	CHP Capacity (MW)
Bubbling Fluidized Bed	131.5	0.0
Circulating Fluidized Bed	147.0	0.0
Downdraft Gasifier	0.1	0.1
Stoker - Grate	315.0	140.5
Suspension Fired Boiler	25.0	0.0
Not specified	19.0	19.0
Total	637.6	154.6

Municipal solid waste (MSW) constitutes the second major contributor to total biomass in California. The main process for disposal of MSW in the state is accomplished by landfills. The assembly bill AB939 required a diversion of 50% of all potential MSW by the year 2000, and more recently assembly bill AB341 was passed to achieve 75% recycling of all waste including organic material by the year 2020, and AB1826 was specifically targeted to increase the diversion of organic waste and hence reducing the amount of waste sent to landfills. Before AB341 and AB1826 were passed, the CBC estimated that a capacity of 1690 MW could be met by landfill gas from MSW. The implementation of these new assembly bills will likely reduce the amount of biodegradable waste reaching landfills, and as a result, reducing the capacity for long-term production of landfill gas.

The location of major landfills is generally in the outskirts of highly populated areas. Thus, in California, the largest landfills are around the Los Angeles metropolitan area, San Diego, and the Bay Area. Figure 2 presents the technical potential for landfill-gas-to-power installations in the year 2020 and the location of the existing facilities. Currently, the total capacity of biopower generated in landfills is 371 MW, which is 22% of the estimated technical potential in California. Table 2 presents the technology distribution in landfill gas biopower installations. The largest fraction of biopower is generated by gas turbines and reciprocating engines. Typically, the heat demand in landfills and surroundings is low, which disincentivizes installation of combined heat and power plants.

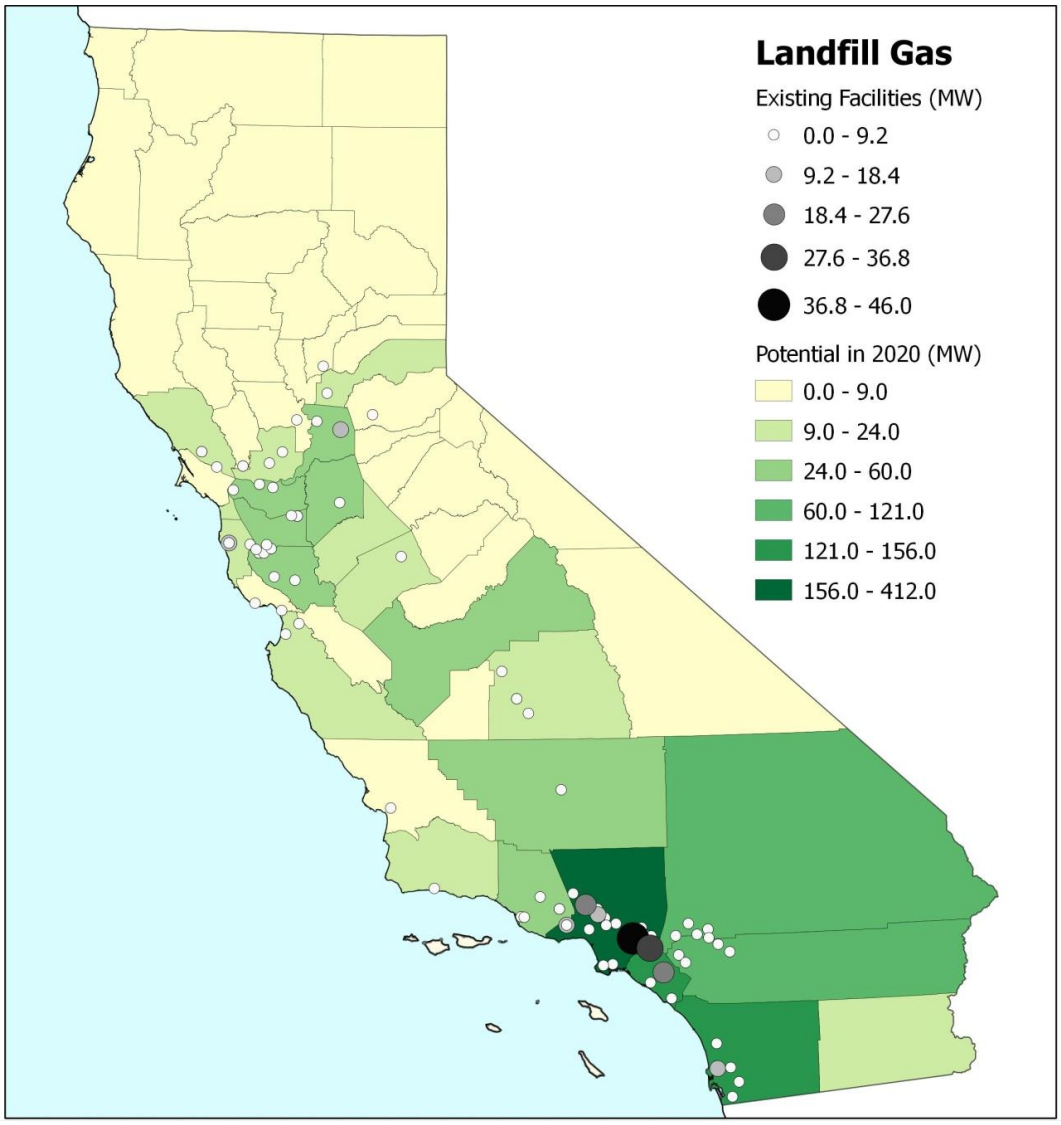


Figure 2: Landfill gas potential for biopower production in 2020 and capacity and location of existing facilities in California.

Table 2: Technology distribution for landfill gas biopower installations

Technology	**Net Capacity (MW)**
Gas and Steam Turbines	11.7
Gas Turbine	116.2
Microturbine	12.0
Reciprocating Engine	173.4
Steam Turbine	58.0
Total	371.3

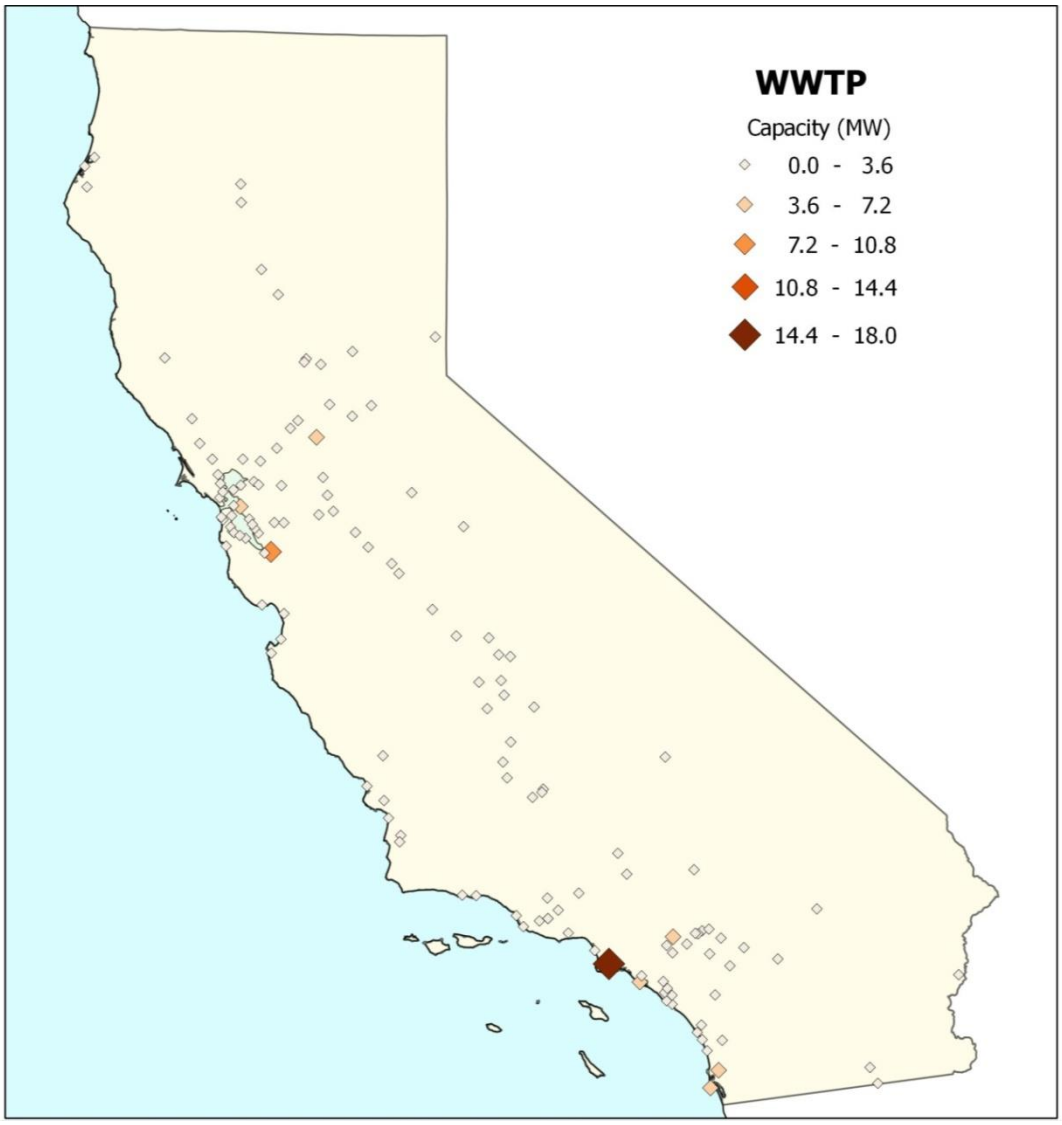


Figure 3: Capacity and location of existing biopower facilities in California in wastewater treatment plants (WWTP).

Table 3: Technology distribution in biopower installations in wastewater treatment plants

Technology	Net Capacity (MW)
Fuel Cells	3.3
Boilers	1.8
Microturbine	1.3
Pipeline	0.4
Reciprocating Engine	43.8
Gas Turbine	18.0
Total	68.6

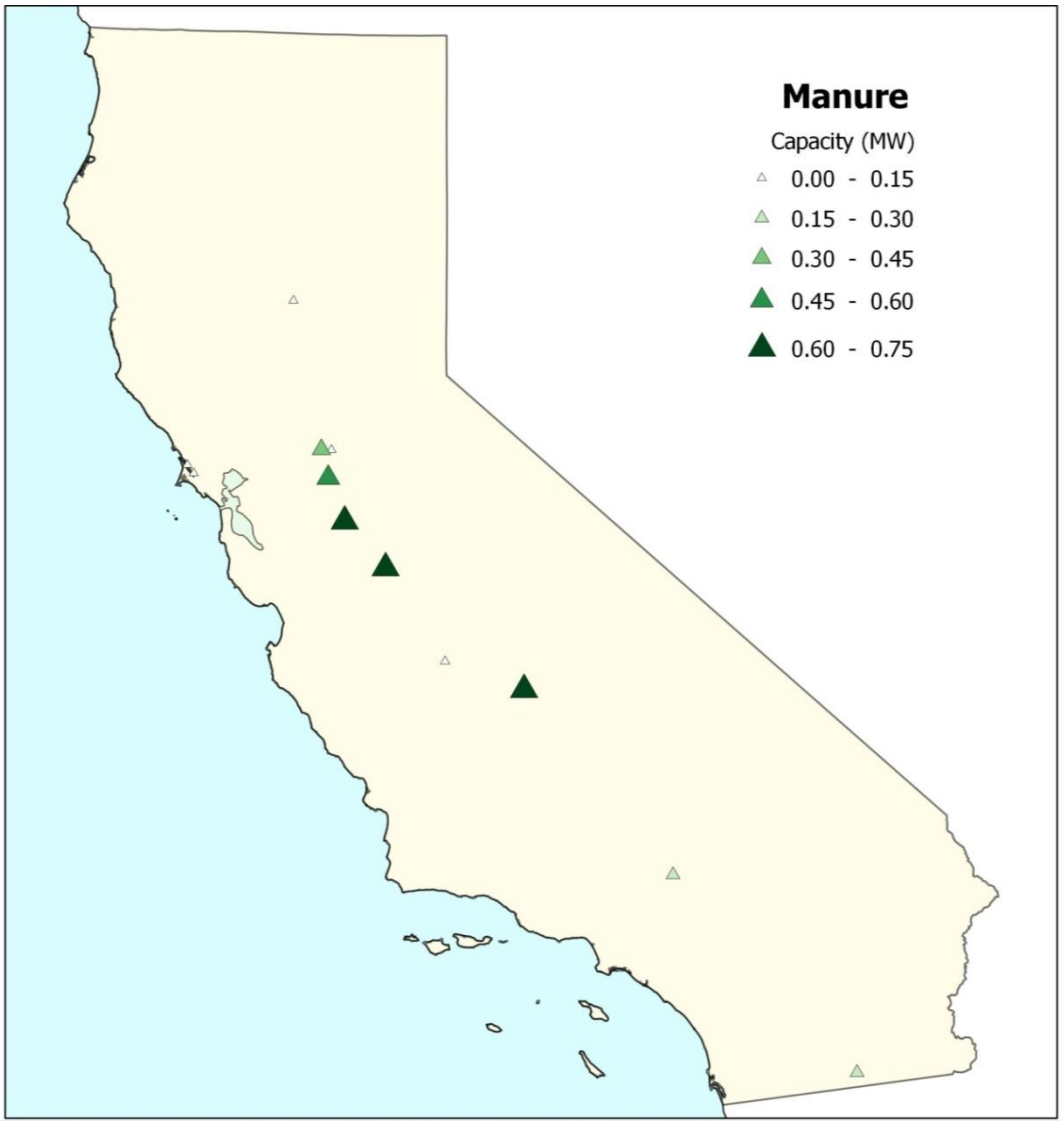


Figure 4: Capacity of existing biopower facilities in California using biogas from animal manure.

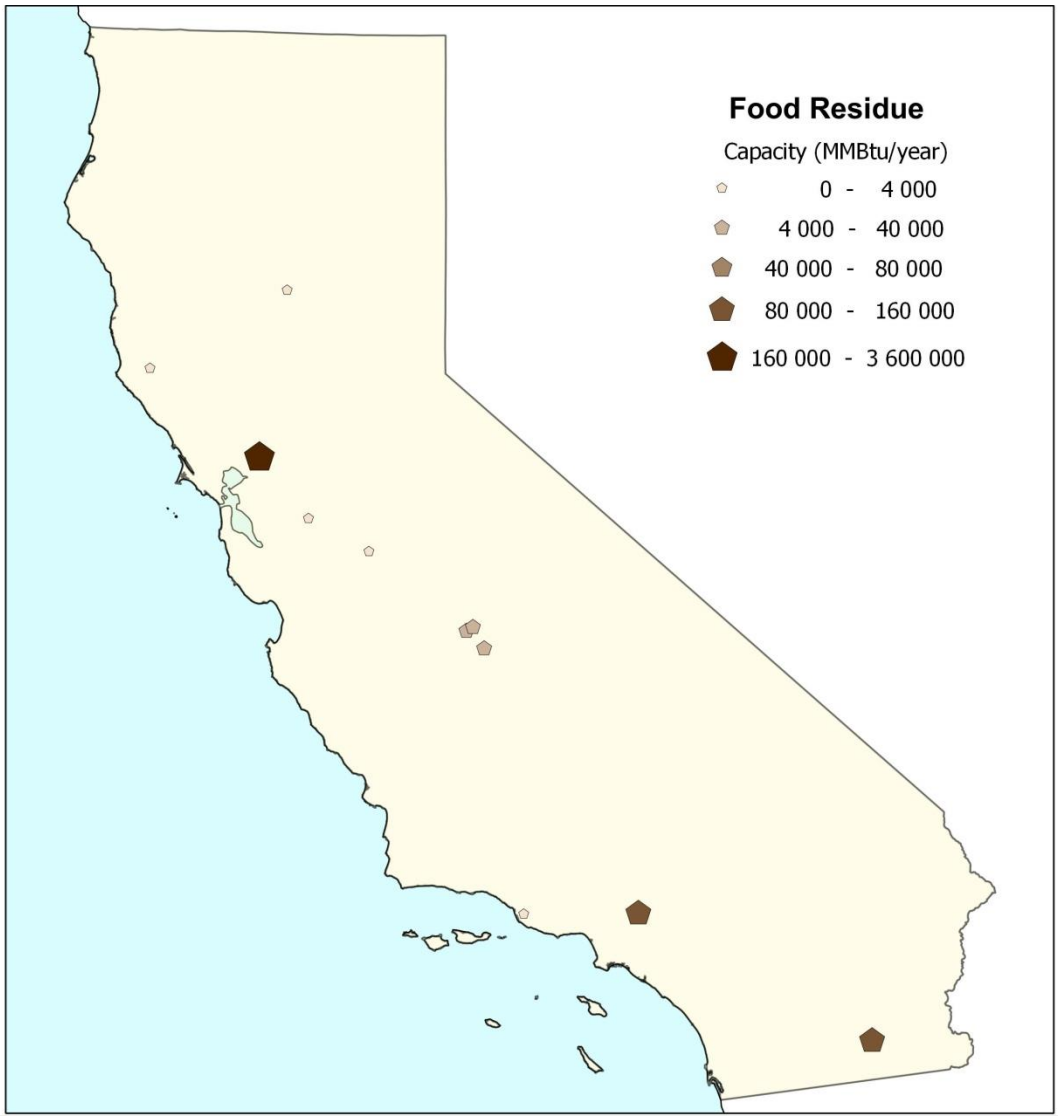


Figure 5: Capacity and location of existing biogas facilities in California from anaerobic digestion of food residue.

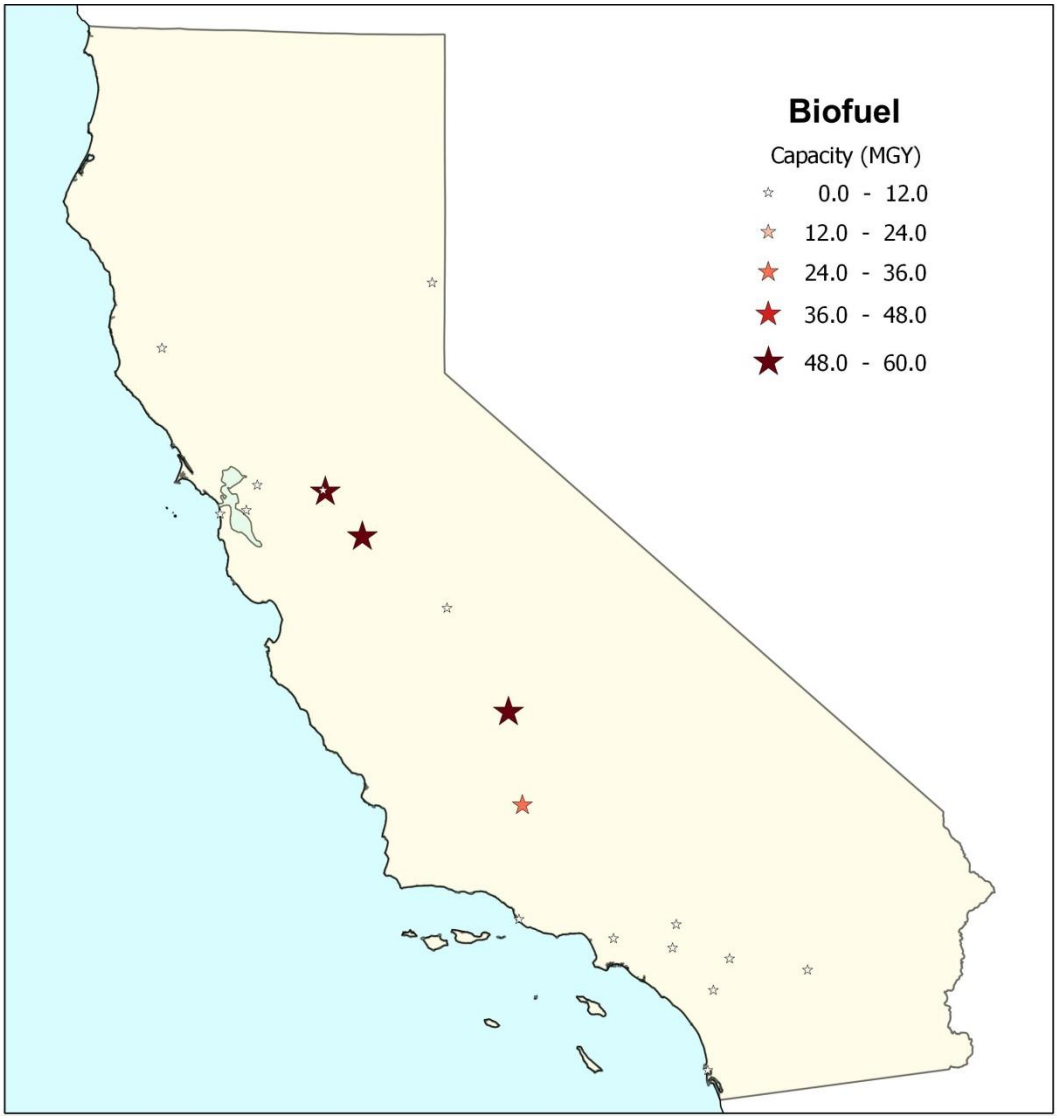


Figure 6: Capacity and location of existing biofuel facilities in California

3 Uses of Biomass

3.1 Biopower

Generation of electricity from biomass is unique among the potential technologies for meeting RPS goals in that it is associated with the generation of substantial amounts of GHGs and pollutants at generation sites during operation. This feature elucidates the importance in assessing GHG and air quality impacts from biopower.

Biomass can be defined as all matter from living and dead biological systems, but when discussing renewable energy sources, it is typically defined as matter from living or *recently* living biological systems. Biomass fueled power plants provided 2.1% and 2.4% of California's total electricity needs and 19.3% and 17.5% of the total renewable electricity generated in 2007 and 2010 (CEC, 2010). California Executive Order S-06-06 requires 20% of the renewable

electricity generated in California to come from biopower resources in 2010 through 2020. The biopower percentage of total renewable electricity generated has declined, and the 2011 Bioenergy Action plan prepared by the California Energy Commission addresses the issues impeding biopower expansion in the state and provides recommendations to increase new installations, prevent idling of current installations, and restarting of idle plants. Williams et al. projected that the technically recoverable biomass from waste and residue streams in 2020 could provide 11.9% of California's electricity needs in 2020 (Williams et al., 2007). This could significantly contribute to meeting California's Renewable Portfolio Standard goals of 33.3% renewable energy contribution to the state's electricity needs in 2020 as well as also reducing greenhouse gas emissions associated with these waste/residue streams. However, significant expansion of biomass facilities in order to reduce greenhouse gas (GHG) emissions could lead to increased environmental stresses without proper analysis and planning, e.g., direct combustion of woody biomass to generate electricity may significantly increase pollutant emissions compared to natural gas combined cycle plants. Additionally, poor planning with regard to dedicated energy crops could also lead to increased GHG emissions or only marginal reduction in GHG emissions while also possibly having detrimental environmental impacts on the land, water, and air quality. Therefore, it is important to assess the environmental impacts throughout the life cycle of the particular feedstock and electricity conversion technology employed. The following sections will first discuss the feedstocks available within California and then move into the characterization of the various biomass electricity conversion technologies. Finally, some environmental impacts that have been shown to result from the production of electricity from biomass will be reviewed, although previous work has shown the importance of performing these life cycle assessments for each considered installation such that the many locations and technology specific parameters are used in the assessment; rather than relying on previous studies that may have used more general figures for model parameters.

3.1.1 Feedstock

The biomass resources available within California are categorized in the following manner by Williams et al. (2007).

- Agricultural residue
- Forestry residue
- Municipal solid waste
- Landfill gas
- Sewage digester gas
- Dedicated crops

Figure 7 shows the technically available and existing biomass electricity capacity by feedstock in 2007 as determined in a California Energy Commission study by Williams et al. (2007). The technically available capacity was estimated using several general assumptions relative to the efficiency of the biomass to energy conversion process. There is potential for a large expansion of electricity generation via biomass waste and residue feedstocks. There may be an even larger potential if dedicated energy crops are considered although Williams et al. projects only modest increases in the technical availability of dedicated energy crop expansion within the state (2% of the state's electrical energy needs in 2020 met from technically available dedicated crops).

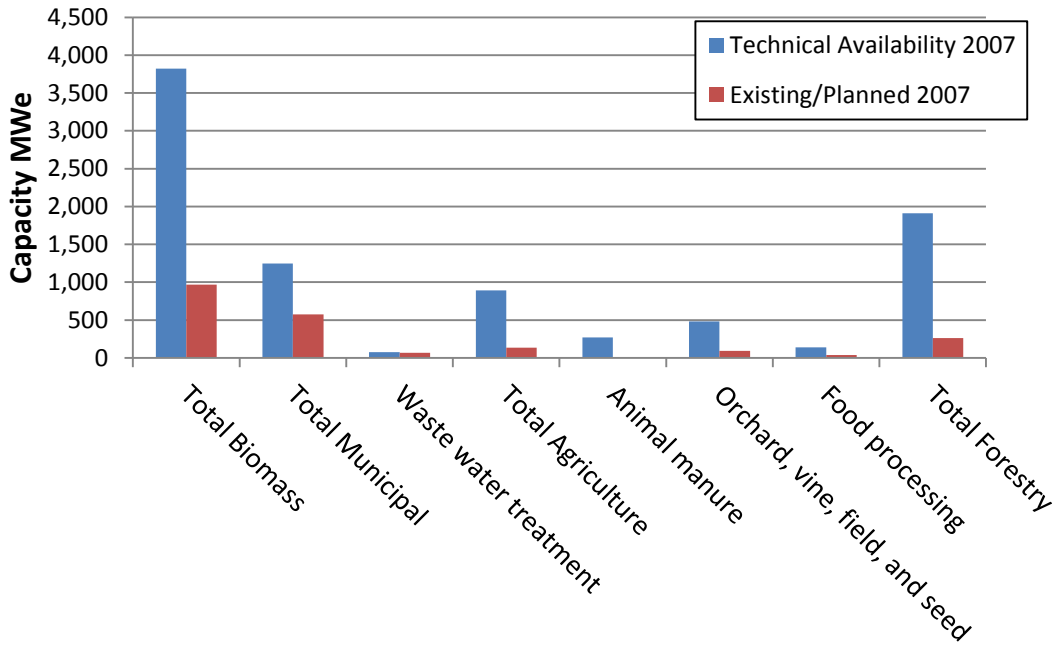


Figure 7: Allocation of biomass resources in California (Williams et al., 2007)

The utilization of waste/residue streams can contribute to GHG emission reductions since the decomposition or treatment of these waste/residue streams result in GHG emissions, which in some cases may be emissions of high global warming potential methane. Forestry residues represent the largest potential for generating electricity from biomass waste/residue available in the state (Figure 7). Existing capacity that uses forestry residues as fuel typically burn the biomass directly to generate steam to drive a turbine which is the same process used by many coal power plants in the US. Pollutant emissions from these direct combustion plants typically exceed those of natural gas fired plants, which may have significant air quality impacts. Additional potential impacts include soil quality and water quality impacts that result from the removal of these residues which would otherwise have decomposed in place. Large expansion in the use of agricultural residues and municipal solid waste (waste water treatment resources are already highly utilized via anaerobic digestion methods) are also possible. Most of the existing capacity for agricultural residue is in the form of direct combustion, which, in a similar manner to the direct combustion of forestry residue, has air quality implications. The treatment of animal manure using anaerobic digestion can contribute nicely to GHG emission reductions but the current use of the digester gas in economically viable heat engines (reciprocating, gas turbines) will not meet current pollutant emission regulation. This is a result of the poor air quality in the regions where animal manure is produced (San Joaquin Valley). Implementation of cleaner technologies such as fuel cells would meet pollutant emission standards but these cleaner technologies remain expensive. Landfill gas utilization is an example where GHG emission reductions have been made via the installation of a large amount of existing capacity as a result of regulations regarding landfill gas emissions and their recovery for flaring or energy use Weitz et al. (2002). However, the use of landfills is being phased out in certain parts of the world such as Europe (EC, 2001). In these locations, the controlled anaerobic digestion of the organic fraction of municipal solid waste (OFMSW) in bioreactors as well as incineration of the

OFMSW is being used for the management of this waste (gasification is also being considered in some instances). The motivation for this are limited land resources and the adverse environmental effects of landfilling such as the leakage of landfill gas (high global warming potential) due to the inability of the wells to capture this gas with 100% efficiency (USEPA, 1995). Leakage of leachate in landfills can also contaminate groundwater. The various environmental impacts associated with biomass power generation are potentially significant especially with regard to the pollutant emissions from those direct combustion conversion technologies that are the most widespread. Although air quality impacts can be substantial, other impacts that are important when considering biomass resources are soil quality, water quality, and biodiversity impacts that might occur as a result of harvesting residues. These environmental impacts will be discussed in more depth in a subsequent section but prior to that discussion the various technologies used in the conversion of biomass into electricity will be characterized more fully.

3.1.2 Electricity Conversion Technologies

Biomass conversion methods can be categorized as follows: thermal, biological, and mechanical. Thermal conversion is currently the method by which most of the biomass generated electricity (biopower) is produced in the US and CA (Williams et al, 2007; Boundy et al., 2011). Figure 8 illustrates the different processing and conversion methods and the various corresponding products. It is important to note that some of the conversion pathways allow for co-products that may have beneficial synergistic effects on the overall system efficiency (Bridgwater, 2006). For direct combustion systems, biomass is burned directly to generate heat for use in a Rankine (steam) cycle rather than converting the biomass to another fuel before combustion. Digestion refers to a process wherein the biomass is digested using bacteria in oxygen deficient (anaerobic) conditions to produce a digester gas and solid digestate. This process occurs in landfills in an uncontrolled manner, and in this application the gas produced is called landfill gas rather than digester gas. Anaerobic digestion is widely used in waste water treatment plants for the processing of this waste stream. The digester gas produced in these plants is also widely used to generate electricity as seen in Figure 7. Anaerobic digestion may also be used to process the organic fraction of municipal solid waste, which is currently utilized to some degree in Europe, however, the solid content of these waste streams must still be below 40% or diluted with water to 40% solids content (EC, 2001; Vandervivere et al., 2003). Gasification is a thermal process where the solid biomass is converted to gas by heating the solid biomass in a manner that produces a gas instead of full combustion. Gasification technologies may provide benefits in efficiency and lower pollutant emissions, however, this technology is not yet fully commercial (Bridgwater, 2006). Pyrolysis is another thermal process and is actually the first step in a gasification process, however, in pyrolysis only this first step is completed yielding a different product that contains volatile liquids and gases. Given that the focus of this report is on renewable electricity generation the processes producing transport fuels will not be considered here, i.e., fermentation and mechanical processes (See Figure 8).

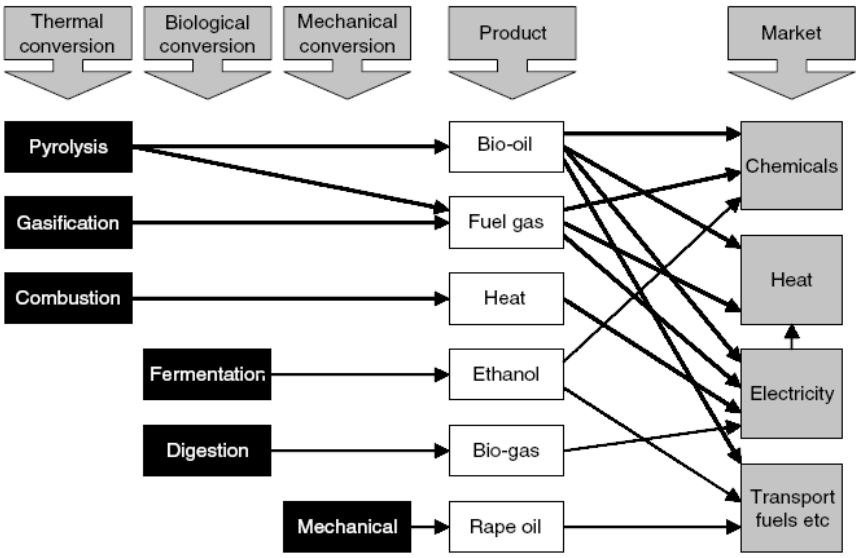


Figure 8: Different biomass conversion technologies and the associated potential products (Brusstar et al. 2005)

3.1.2.1 Direct Combustion

The direct combustion of biomass in boilers for steam production in Rankine cycles is a fully commercialized technology with many plants in California that have been in operation for 20 years or more (See the National Electricity Energy Data System) (EPA, 2006). This technology is most commonly used in the conversion of solid biomass although it could also be used for the conversion of biogas or syngas, it is typically not done since the use of the gaseous fuel in another thermodynamic cycle produces higher efficiencies. This section will focus on the different types of boilers currently used to burn solid biomass. The most frequently used boilers in these systems are stoker and fluidized bed boilers (EPA, 2007), but pulverized fuel boilers will also be discussed here.

3.1.2.1.1 Stoker Boilers

Stoker boilers were first introduced in the 1920s for use with coal (EPA, 2007). Combustion air is fed from under the grate upon which the solid fuel burns. This grate can either move or remain stationary but must allow for the removal of ash. Air is usually also injected at locations above the grate to ensure complete combustion (overfire air). The air flow design is very important in biomass stoker boilers for efficient and complete combustion with typical modern biomass designs having more overfire air than in coal systems with air flow splits between the overfire and underfire flows being 60% and 40%, respectively (EPA, 2007). The manner in which the fuel is distributed over the grate is a major mode of classification. The fuel can be fed onto this grate from underneath the grate (underfeed) or over the grate (overfeed). Underfeed stokers are usually best suited for dry fuels (i.e., less than 40-45% moisture content) and are less popular because of their higher cost and worse environmental performance compared to overfeed stokers (EPA, 2007). Overfeed stokers can be further classified into mass feed and spreader categories. Again these names refer to how the fuel is distributed over the grate. Mass feed stokers typically feed fuel into the furnace at one end and use a moving grate to distribute the fuel throughout the furnace. Spreader stokers will actually throw the fuel into the furnace above the grate such that the fuel is distributed evenly across the grate which allows for more even air flow distribution

throughout. This "throwing" is done using air injection or overthrow/underthrow rotors. This also results in more suspension burning in these boilers, which results in better response times compared to mass fed or underfed boilers (EPA, 2007). Spreader stokers are the most common stoker boilers (EPA, 2007).

3.1.2.1.2 Fluidized Bed Boilers

Fluidized beds were initially studied by Winkler in the 1920s for application as a gasifier, and in the early 1960s the US and UK began programs focused on this technology for the development of a compact boiler package that could reduce costs. These early studies showed that emissions could also be reduced by utilizing this technology (Highley, 1980). Since then with certain governmental regulations and funding opportunities, this technology has become commercial with every major US boiler manufacturer offering an atmospheric fluidized bed combustor in their product line (DOE, 2006). However, the more advanced technologies (pressurized and supercritical fluidized beds) have only several units operational (six-pressurized; 1-supercritical) and are in need of additional research and development due to their potential for higher efficiencies compared to the older commercially available atmospheric technologies (Koomneef et al., 2007; Patel, 2009). Fluidized bed boilers burn fuel in a fluidized state, i.e., in a bed of granular solids with typical sizes 0.1 to 1 mm (depending on the boiler type) with primary combustion air flowing up through the bed material where the temperature of this bed material is typically maintained at 800-900 °C through heat transfer either to the flue gas or heat exchange tubes buried in the bed material (Basu, 2006). This lower operating temperature compared to that of the stoker boilers results in lower NOx production. The bed material can be sand, gravel, limestone, ash, or other special synthetic materials. The interaction of the bed material with the fuel as it is burning allows for more efficient combustion as well as the ability to capture pollutants (e.g., addition of limestone absorbs SOx).

The two main types of fluidized bed boilers are the bubbling fluidized bed (BFB) and the circulating fluidized bed (CFB) with further classification according to operating pressure (atmospheric vs. pressurized) and state of the steam product (sub vs. supercritical). The BFB technology was first to become commercial with the CFB becoming commercial later. There are now more CFB units in operation than BFB units (Koomneef et al., 2007). The velocity of the primary air flowing through the bed is higher in the CFB than in that BFB, which is the primary distinction between these two technologies. The CFB primary air flow is high enough to actually blow the bed material upward to the top of the furnace where it is then separated from the flue gas and re-circulated to the bottom of the furnace. The more advanced technologies attempt to increase the efficiency of these systems by increasing the operating pressure for combined cycle operation or by increasing the temperature and/or pressure of the steam produced to supercritical conditions. Each of these methods of increasing efficiency can be applied to the BFB or CFB technologies although the CFB technology is typically used because of the higher combustion efficiencies and better sulfur capture achievable with these systems compared to the BFB (Koomneef et al., 2007; Basu, 2006).

3.1.2.1.3 Pulverized Fuel Boilers

Pulverized fuel boilers are less likely to be used for biomass combustion; although co-firing pulverized coal plants with biomass has been accomplished. This is because of the much more intensive processing of the biomass prior to combustion, i.e., to attain the appropriate particle sizes (<10mm) (Van Loo, 2008). However, higher efficiencies are achievable with these systems

when compared to BFB and CFB technologies because of the lower excess air used (See Table 4). But during the bidding process of a supercritical CFB in Poland, it was found that the CFB option was 20% cheaper in capital cost and 0.3% higher in net efficiency than the competing supercritical pulverized coal option (Basu, 2006). Additionally, these systems require post processing for SOx removal unlike the fluidized bed options.

3.1.2.1.4 Summary of Issues Related to Direct Combustion of Biomass

Table 4 summarizes the advantages and disadvantages of several direct combustion technologies.

Table 4: Summary of the advantages and disadvantages of various direct combustion technologies (Van Loo, 2008)

Advantages	Disadvantages
Grate Furnaces	
• Low investment costs for plants <20MWth • Low operating costs • Low dust load in flue gas • Less sensitive to slagging than fluidized beds	• Usually no mixing of wood fuels and herbaceous fuels possible (only special constructions can cope with such fuel mixtures) • Efficient NOx reduction requires special technologies (combination of primary and secondary measures) • High excess oxygen (5-8% vol) decreases efficiency • Combustion conditions not as homogeneous as in fluidized beds • Low emission levels at partial load operation requires a sophisticated process control
Underfeed stokers	
• Low investment costs for plants <6MWth • Simple and good load control due to continuous fuel feeding and low fuel mass in the furnace • Low emissions at partial load operation due to good fuel dosing • Low flexibility in regard to particle size	• Suitable only for biomass fuels with low ash content and high ash melting point (wood fuels) (<50 mm)
BFB furnaces	
• No moving parts in hot combustion chamber • NOx reduction by air staging works well • High flexibility concerning moisture content and kind of biomass fuels used • Low excess oxygen (3-4%) raises	• High investment costs, interesting only for >20MWth • High operating costs • Reduced flexibility with regard to particle size <80mm • Utilization of high alkali biomass fuels (e.g., straw) is critical due to possible bed

efficiency and decreases flue gas flow	agglomeration without special measures • High dust load in the flue gas • Loss of bed material with the ash without special measures
CFB furnaces	
• No moving parts in the hot combustion chamber • NOx reduction by air staging works well • High flexibility regarding moisture content and kind of biomass fuels used • Homogeneous combustion conditions in the furnace if several fuel injectors are used • High specific heat transfer capacity due to high turbulence • Use of additives easy • Very low excess oxygen (1-2%) raises efficiency and decreases flue gas flow	• High investment costs, interesting only for plants >30MWth • High operating costs • Low flexibility with regard to particle size (,40mm) • Utilization of high alkali biomass fuels (e.g., straw) is critical due to possible bed agglomeration • High dust load in flue gas • Loss of bed material with the ash without special measures • High sensitivity concerning ash slagging
Pulverized fuel	
• Low excess oxygen increases efficiency (4-6%) • High NOx reduction by efficient air staging and mixing possible if cyclone or vortex burners are used • Very good load control and fast alteration of load possible	• Particle size of biomass is limited (<10-20mm) • High wear rate of the insulation brickwork if cyclone or vortex burners are used • An extra start up burner is necessary

3.1.2.2 Gasification

Gasification technologies are less available commercially than direct combustion technologies, however, gasification provides opportunities for cleaner plant operation and higher efficiencies (EPA, 2007). This process is different from direct combustion in that the solid fuel is partially oxidized in an oxygen deprived environment sometimes with the addition of steam or carbon dioxide such that a gas is produced. This gas has a low heat content (5000-15000 kJ/kg) and the remaining char may still have a heating value associated with it which results in less than 100% energy conversion from the original solid fuel (typical conversion efficiencies are 60-80% (EPA, 2007). The process of gasification occurs in four sets of processes: drying, pyrolysis (devolatilizaton), combustion, and reduction (Basu, 2006). The first, second, and last of these processes are endothermic, absorbing heat from the combustion process. The drying process occurs quickly (>150 °C) with pyrolysis reactions following this process (150-700 °C). The pyrolysis process is complex and progresses to fast reaction rates at higher temperatures. The

pyrolysis process is responsible for the production of some gases, tar, and char. Tar causes many issues in gasification processes (Knoef, 2000). The combustion process occurs in an oxygen deprived atmosphere thereby only partially oxidizing the solid fuel rather than completing the combustion process. These partial oxidation reactions supply the heat required for the endothermic processes (i.e., drying, pyrolysis, reduction). The process of reduction or gasification involves several main sets of reactions: the water gas, Boudouard, water gas shift, and methanation reactions (Basu, 2006).

Gasification units are classified according to the oxidant used (oxygen vs air blown gasifiers) and according to the reactor technology used (fixed/moving bed, fluidized bed, entrained flow). The typical efficiencies and example schematics of these systems are shown in Figure 9 and Figure 10, respectively. The fixed bed gasifiers can be further classified by the flow of the gasifying medium (air/steam/oxygen): updraft, downdraft, side draft/cross flow. Fluidized bed gasifiers are classified in a similar fashion to fluidized bed boilers/combustors (i.e., circulating vs. bubbling, atmospheric vs. pressurized). The commercial availability of each technology was inventoried in 2000 for the European Commission through industry surveys (Knoef, 2000). This inventory showed that downdraft gasifiers accounted for 75% of commercially available products with fluidized beds accounting for 20%, updraft for 2.5% and 2.5% of other types (Bridgwater, 2006).

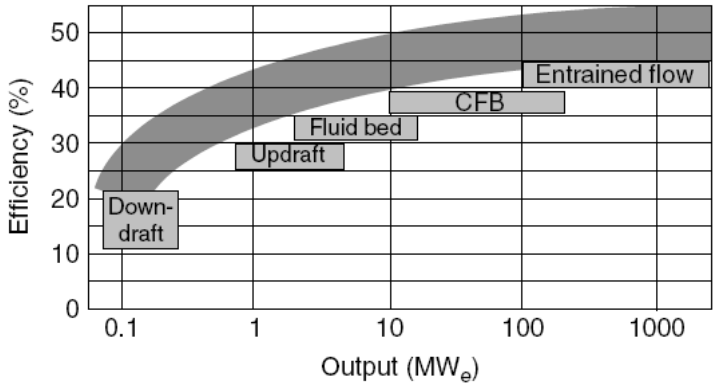


Figure 9: Typical electrical conversion efficiencies for different types of gasification technologies (Bridgwater, 2006)

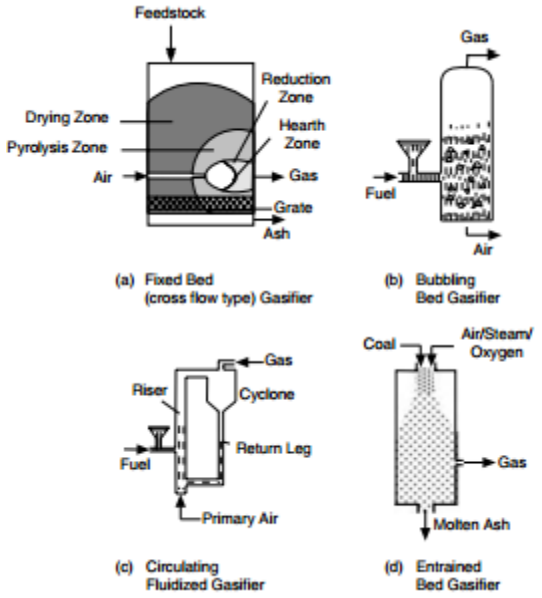


Figure 10: Schematic representations of different types of gasifiers (West et al., 2009)

3.1.2.2.1 Fixed/Moving Bed Gasifiers

In the fixed/moving bed design, the solid fuel is fed into the bed while the gasifying medium (i.e., steam, air, or oxygen) flows past the fuel. The flow of this gasifying medium is how these designs are classified: updraft, downdraft, and side draft/cross flow. In the case of an updraft gasifier, the gasifying medium feed flows upward through the bed of fuel, char, and ash as seen in Figure 11 with different reactions occurring in the bed. Fixed bed gasifiers are limited to small scale applications typically less than 2-5 MW (Bridgwater, 2006; EPA, 2007).

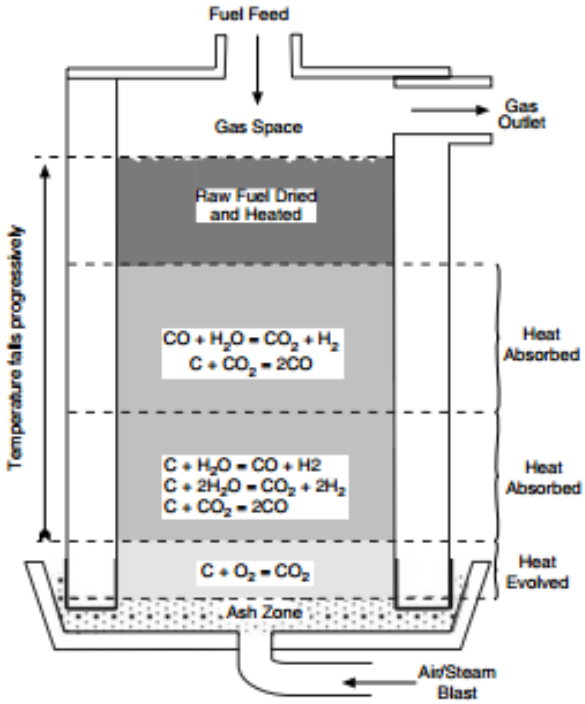


Figure 11: Schematic of an updraft gasifier, taken from Basu, 2006

3.1.2.2.2 Fluidized Bed Gasifiers

Fluidized bed gasifiers were first studied in the 1920s by Winkler, and in fact he developed a commercial air blown fluidized bed gasifier (EPA, 2007; Basu, 2006). The fluidization of the bed is completed in a similar fashion to those in fluidized bed boilers, however, the fluidization is accomplished by the gasifying medium which can be air, steam, or oxygen. As in the case of fluidized bed boilers, fluidized bed gasifiers can have bubbling (BFB) or circulating fluidized beds (CFB) operating at either pressurized or atmospheric conditions. BFB gasifiers have lower gasifying medium velocities compared to CFB gasifiers where the gasifying medium flow rate is high enough to actually blow the bed material upward to the top of the gasifier where the bed material is then separated from the syngas and circulated back to the bottom of the gasifier. Similar to the fluidized bed boilers except that the product is now a synthetic gas (syngas) rather than a hot flue gas for producing steam.

3.1.2.2.3 Entrained Bed Gasifiers

Entrained flow systems require pulverized fuel particles to be used (<0.15 mm). These fuel particles are typically injected at the top of the gasifier along with the gasifying medium, and these particles are surrounded/suspended/entrained by the gasifying medium. These gasifiers are usually used in coal gasification processes for large systems (>100MWe). Biomass gasification with this technology is not typical because of the fuel particle size requirement. However, the syngas produced has very low or zero tar content in addition to high carbon conversion efficiencies.

3.1.2.2.4 Hybrid or Other Gasification Technologies

There are other gasification technologies that may have hybridized two technologies; may have slightly different reactor conditions such that the technology does not fit neatly into the classifications given here; or the technology could be completely different. One example of a hybridized approach is the Gussing gasifier in Austria that uses a dual fluidized bed process wherein one bed operates in a combustion mode which supplies heat to the other bed which operates in a gasification mode. Other twin fluidized bed gasifiers have been investigated in Europe and Asia (Corella et al., 2007). This gasification process has also been termed indirect gasification and has been quite successful (Bridgwater, 2006; Thunman et al., 2010). Another example of a different gasification technology is plasma gasification where a plasma torch (electric arc between two electrodes) is used to provide the heat for the gasification process. This technology requires electricity but it is insensitive to the feedstock type (Basu, 2010).

3.1.2.2.5 Summary of Issues Related to Gasification

Table 5 summarizes the various issues related to each gasification technology discussed above.

Table 5: Summary of challenges and advantages of the various gasification technologies (compiled from (Bridgwater, 2006; Basu, 2010; Wang et al., 2008)

	Main Advantages	Main Technical Challenges
Gasifying Agents		
Air	1. Partial Combustion for heat supply of gasification 2. Moderate char and tar content	1. Low heating value (3-6 MJ/Nm3) 2. Large amount of N2 in syngas 3. Difficult determination of

		equivalence ratio
Steam	1. High heating value (10-15 MJ/Nm^3) 2. H2 rich syngas	1. Require indirect or external heat supply 2. Required catalytic tar reforming
Carbon Dioxide	1. High heating value syngas 2. High H2 and CO in syngas and low CO2 in syngas	1. Require indirect or external heat supply 2. Required catalytic tar reforming
Oxygen	1. High heating value syngas (12-28 MJ/Nm^3) 2. Higher quality gas (low tar)	1. Energy intensive to supply oxygen 2. Expensive
Gasifier Design		
Fixed/Moving Bed	1. Simple and reliable design 2. Capacity for wet biomass gasification 3. Favorable economics on small scale	1. Long residence time 2. Non uniform temperature distribution 3. High char and/or tar contents 4. Low cold gas energy efficiency 5. Low productivity
Fluidized Bed	1. Short residence time 2. High productivity 3. Uniform temperature distribution 4. Low char and/or tar content 5. High cold gas efficiency 6. Reduced ash related problems	1. High particulate dust in syngas 2. Favorable economics on medium to large scale
Gasifier operation		
Increase of temperature	1. Decreased char and tar content 2. Decreased methane in syngas 3. Increased carbon conversion 4. Increased heating value of syngas	1. Decreased energy efficiency 2. Increased ash related problems
Increase of pressure	1. Low char and tar content 2. No costly syngas compression required for downstream utilization of syngas	1. Limited design and operational experience 2. Higher costs at small scale
Increase of equivalence ratio	1. Low char and tar content	1. Decreased heating value

3.1.2.3 Pyrolysis

Pyrolysis is defined as thermal decomposition in the absence of oxygen and is the first step in combustion and gasification processes. This process of pyrolysis can be performed in different modes as seen in Table 6. Pyrolysis has been proposed for the production of bio-oils and given that fast pyrolysis provides the highest yield of liquids this is the typical mode of operation for the production of bio-oils from pyrolysis (Bridgwater, 2006).

Table 6: Typical product yields obtained from different modes of pyrolysis of dry wood (Bridgwater, 2006)

Mode	Conditions	Liquid (%)	Char (%)	Gas (%)
Fast	Moderate temperature, around 500 °C, Short hot vapour residence time, ~1 s	75	12	13
Intermediate	Moderate temperature, around 500 °C, Moderate hot vapour residence time ~10–20 s	50	20	30
Slow (carbonisation)	Low temperature, around 400 °C, very long residence time	30	35	35
Gasification	High temperature, around 800 °C, long residence times	5	10	85

The different pyrolysis reactors (pyrolysers) are fluidized beds (CFB and BFB), transported bed, entrained bed, and ablative. The fluidized and entrained beds are similar to the reactors used in the boiler and gasification processes but with different residence times and reactor temperatures. The ablative pyrolyser mechanically applies pressure to the biomass particles such that an appropriate rate of heat transfer is achieved (biomass particles can be larger in this reactor than in the others where small particles are required for sufficient heat transfer). Bridgwater likened this process to the melting of butter in a frying pan (Bridgwater, 2006). The pyrolysis oil must then be collected and in fact the reactor only amounts to about 10-15% of the total plant cost yet most of the research has been focused on this part of the process. Figure 12 shows a conceptual schematic for a pyrolysis plant being utilized for the production of bio-oil (Bridgwater, 2006).

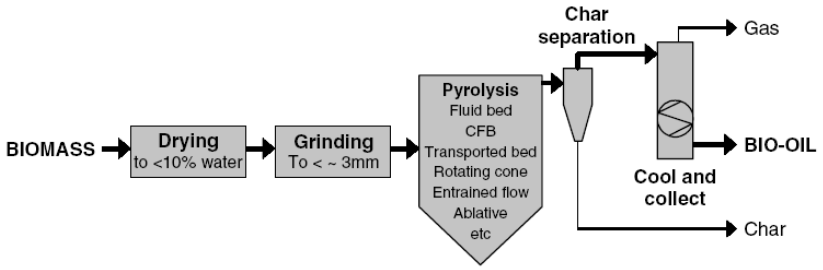


Figure 12: Schematic of a fast pyrolysis process (Bridgwater, 2006)

Charcoal and gas are by-products of fast pyrolysis, and they typically contain 25 and 5%, respectively, of the energy in the biomass feedstock. Some of these byproducts must be utilized in the pyrolyser to supply heat. The bio-oil produced would ideally be readily used as a substitute for conventional liquid fossil fuels, however, differences in properties prohibit easy substitution, which is not to say that it cannot be done.

3.1.2.4 Digestion

Anaerobic digestion is the conversion of organic matter using certain types of bacteria in the absence of oxygen. This process produces a fuel gas with a methane content of 50-80% with the balance being mostly CO_2 in addition to small amounts of hydrogen sulfide, nitrogen, hydrogen, methylmercaptans, and oxygen. Residue slurry called digestate is also produced in this process.

Aerobic digestion is another process of conversion of organic matter, however, this occurs in the presence of oxygen with the major products being compost, carbon dioxide, and water. Since this does not provide a fuel gas, it is not considered for bio-energy applications although it is used for processing of waste in some landfills. The process of anaerobic digestion is used in both anaerobic digesters (controlled) and in landfills (uncontrolled) (Basu, 2010), but typically, the term anaerobic digestion is used when referring to anaerobic digesters and not landfills. The anaerobic digestion of organic matter consists of several steps: hydrolysis, fermentation, acetogenesis (Beta-oxidation), and methanogenesis (Nayono, 2009). Figure 13 illustrates these steps schematically.

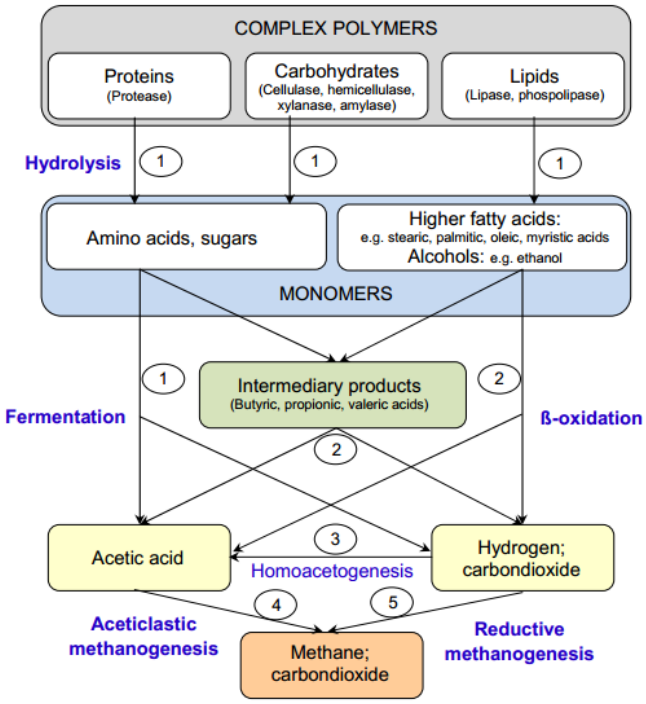


Figure 13: Illustration of the various sets of biological reactions that occur in anaerobic digestion (U.S. EPA, 2010)

3.1.2.4.1 Anaerobic Digesters

Anaerobic digesters are classified according to the digester temperature (psychrophilic, mesophilic, thermophilic), feed mode (batch vs. continuous), and solids content in feed (i.e., wet vs. dry). They have traditionally been used for processing of wet waste (<15% solid content), but new developments in solid state fermentation have allowed higher solid content (Brusstar et al., 2005). The typical temperature ranges for psychrophilic, mesophilic and thermophilic digestion are respectively: 5-20 °C, 30-38 °C, and 50-57 °C. Thermophilic conditions provide higher biogas production, increased solids reduction, improved dewatering, and increased destruction of pathogenic organisms; however, these bacteria have less process stability due to their sensitivity to temperature fluctuations, are more energy intensive, and have a higher odor potential (Appels et al., 2008). Mesophilic conditions in contrast have lower biogas production rates but have better stability. Digestion under psychrophilic conditions is being considered as a low cost alternative because no added heat is required for the feed, although it requires long residence times for digestion due to low temperatures (Saady and Masse, 2013). Figure 14 shows the

regions of temperature for the different bacteria and the corresponding relative rates of reaction. The different feed modes are straightforward to understand. Batch mode is where the digester is filled with waste once and then left to proceed through the digestion process without the addition of more waste. This has sometimes been termed "landfill in a box", however, the biogas production of batch systems is much higher than in landfills because of the active control of the system through recycling of the leachate and operation at higher temperatures than those seen in landfills (Nayono, 2009). The continuous mode is where the waste is continually fed into the digester. Wet digesters are those digesters designed to process waste with a solid content of less than 13% (Vandevivere et al., 2003; Lissens et al., 2001). Batch and continuous systems can also have single or multiple stages where certain processes occur in certain stages, which for example, would allow the separation of hydrolysis and fermentation processes from the methanogenesis process. These multiple stage systems are the most complex and hence the most expensive. Batch systems have the simplest design and least cost. In comparing dry and wet systems, dry designs are more robust and flexible than wet systems. The majority of industrial applications as of 2001 used single stage systems with an even split between dry and wet systems (Lissens et al., 2001).

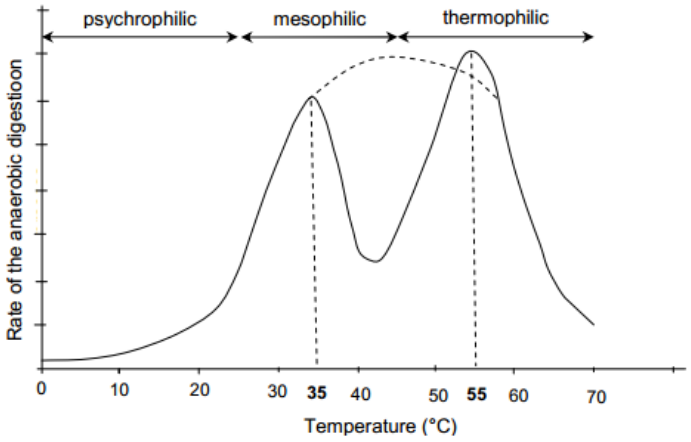


Figure 14: Rate of anaerobic digestion vs. digester temperature (U.S EPA, 2010a)

Further classification is typically applied when discussing digestion of low solid content agricultural solid waste residues, such as manure. Three systems are usually cited as being available to these agricultural enterprises: covered lagoon, complete mix, and plug flow (Demirbas et al., 2005; Krich et al., 2005). Each of these three designs would be classified as wet technologies since they require feeds with less than 13% solids content (Demirbas et al., 2005). The covered lagoon is a specific digester design that requires dilute waste (<2% solids) to be collected in a covered pond or lagoon. The cover allows for the collection of biogas as well as separation from air. These systems are simple and low cost to install, however, they only work well in warm climates since the temperature within the lagoon is not controlled. Complete mix digesters are covered, heated tanks that use a mechanical or gas mixer to keep the solids in suspension. They require a feed with a solid concentration of 3-10%. These units are more complex and expensive than covered lagoons but are suitable for cold climates. Plug flow digesters are also heated and require a feed with a solid concentration of 11-13%. These designs are usually covered for gas collection and rectangular with new feed entering at one end of the digester and the leftover sludge exiting at the other.

3.1.2.4.2 Landfill

The same process of anaerobic digestion occurs in landfills to produce landfill gas; however, landfill processes may be distinguished from digester technologies in that the process is uncontrolled in landfills. Landfill gas is extracted from the sealed landfill through a network of wells drilled in the landfill. However, these wells do not recover the landfill gas with 100% efficiency, rather some leakage still occurs. Typical recovery efficiencies are 60-85% (EPA, 1995). Treatment of the gas coming out through the well head is required. Landfill gas will have a typical methane content of 50-55% (Bridgwater, 2006).

3.1.3 Emissions Impacts

Environmental impacts resulting from the use of biomass for electricity generation (biopower) differ from the environmental impacts of other renewable technologies such as wind and solar in that biopower technologies have operational pollutant emissions comparable to conventional fossil fuel sources, which could potentially have adverse impacts on regional and local air quality. Quantification of GHG benefits is complicated by uncertainty with regards to allocation of any "negative" emissions occurring from carbon uptake. The carbon emissions occurring during the conversion to electricity (typically through combustion) are assumed to be reabsorbed by photosynthesis during re-growth of the biomass, or in the case of the use of waste or residue, would have been emitted during decomposition, therefore, power generation via biomass waste/residue is also considered a carbon offset. Biopower technologies that utilize appropriately selected, dedicated energy crops on the correct land type have the potential to sequester carbon in the soil and crop roots (Tilman et al., 2006; Tolbert et al., 2002). Sequestration technology currently being considered for coal plants can also be applied at biopower plants to effect negative carbon emissions. Additionally, biopower allows dispatch of electricity unlike wind and solar, which are intermittent. Wind and solar must rely on other dispatchable resources to meet unserved load that are typically less efficient, higher emitting fossil fuel technologies. Biopower also has a large environmental impact in terms of land and water resources consumed (See Water Impacts section), especially when considering dedicated energy crops where significant energy inputs occur upstream of the conversion to electricity. Removal of residues such as forestry and agricultural residues may also have an impact on the soil quality and biodiversity (Stewart et al., 2010). There is also the question of transporting the biomass to biopower plant sites, which can also have an environmental impact in terms of pollutant, GHG, and noise emissions as well as traffic congestion. These issues will be discussed in the following sections.

3.1.3.1 Feedstocks

The differences in life cycle analyses of dedicated energy crop and waste/residue feedstocks are important to note because dedicated energy crops require changes in use of land and water resources that affect biodiversity, food resources, hydrologic cycles (Le et al., 2011), surface heat balances (Georgescu et al., 2011), etc. in a complex way that make life cycle environmental impact studies extremely challenging. In fact, varying levels of impacts for the same energy crop species have been claimed by different life cycle assessment studies (Georgescu et al., 2011). The methods of accounting for GHG emissions resulting from land use change have also been questioned (Searchinger et al., 2009). The life cycle GHG emissions are more straightforward when examining biomass residues and wastes where these emissions can be considered as zero (or even negative when methane emissions are mitigated) because decomposition results in emissions irrespective of any energy generation activities. However, there may still be soil

quality, water quality, and biodiversity impacts as a result of residue removal (Stewart et al., 2010). Long term studies investigating the removal of forestry residue in California's mixed conifer forests have concluded that there is no long term loss in forest productivity as a result of residue removal, but similar studies have not been conducted for other forest types or shrublands. Additionally, other benefits and impacts resulting from residue removal have not been quantified such as the possible reduction of wildfire associated emissions and the loss of wildlife habitats (Stewart et al., 2010).

Dedicated energy crops create ecological concern because of possible replacement of food crops, upset of the hydrologic cycle (Le et al., 2011), upset of soil nutrient balance (Adegbidi et al., 2001), biodiversity, effects of land use change on carbon balances (Tolbert et al., 2002; Searchinger et al., 2009), etc. Given these concerns, it should be expected that dedicated energy crops provide more than marginal reductions in GHG emissions when compared to the fossil fuel they are replacing otherwise the risk of these other ecological concerns can be considered too great. Tilman et al. suggest that only several feedstocks be considered: perennial plants grown on degraded lands abandoned from agricultural use, crop residues, sustainably harvested wood and forest residues, double crops/mixed cropping systems, and municipal and industrial wastes (Tilman et al., 2009). Fazio et al. show that the average life cycle GHG emissions are lower for perennial crops than annual crops (Fazio et al., 2011). Adler et al. performed life cycle studies comparing several different energy crops (switchgrass, giant reed, and hybrid poplar) to be used for electricity generation in an integrated gasification combined cycle system (Adler et al., 2007). They showed that the net GHG savings achieved when compared to a coal gasification system were larger than those net GHG savings when used to produce biofuels, which motivates the use of biomass for power generation. Thornley et al. also compared life cycle GHG emissions of short rotation coppice (willow/poplar) to miscanthus for various gasification and combustion systems with some systems including combined heat and power capability (Thornley et al., 2009). Their results show that in terms of the GHG emissions per unit of energy produced, short rotation coppice performs better than miscanthus, however miscanthus performs better in terms of GHG emissions per unit of land used. These results highlight the potential tradeoffs that must be considered with respect to the various available dedicated crops. These researchers also discuss the issue of soil carbon balance, and the dependence upon what the land use was prior to implementation as land for energy crop growth. Thornley et al. also analyzed the life cycle pollutant emissions of the two crops (short rotation coppice and miscanthus) in another publication and found that the biomass production, preparation, and provision was much less significant in determining the CO, NOx, and hydrocarbon emissions than was the electricity production for most of the cases analyzed (Thornley et al., 2008). However, the particulate emissions were largely produced during the biomass production, preparation, and provision phases rather than during the electricity generation phase.

Some researchers such as Tilman et al. (2006) have demonstrated that using low input and high diversity grassland for biopower can actually provide carbon sequestration in the soil and roots of the biomass. However, these demonstrations were in climates much different from California, but some preliminary work has begun in demonstrating the potential of low input grasses (e.g., switchgrass) in California (Pedroso et al., 2011). Appropriate selection of land and feedstock for dedicated energy crop use has high importance in limiting indirect and direct environmental impacts, and the use of thorough life cycle analyses that take into account the specific location of interest are vital to minimizing the impacts.

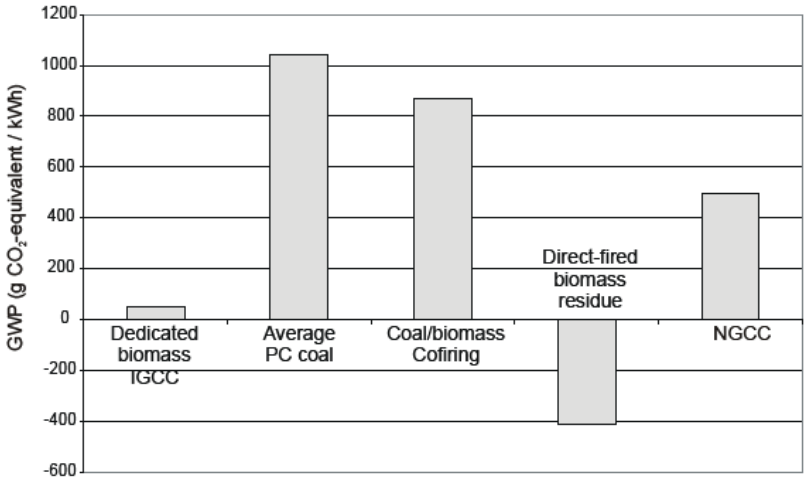


Figure 15: Life cycle GHG emissions for several different scenarios of electricity generation (Bain et al., 2003)

Bain et al. discusses various life cycle assessments performed at the National Renewable Energy Laboratory that illustrate the differences between the use of biomass residues and dedicated energy crops for electricity generation (Bain et al., 2003). The systems considered include a dedicated biomass (hybrid poplar) integrated gasification combined cycle, pulverized coal, coal/biomass co-firing, direct fired biomass residue, and natural gas combined cycle systems. The analyses demonstrate that the use of biomass residue is preferable to the use of dedicated energy crops in terms of both the net energy ratio (energy out /energy in) and the life cycle GHG emissions. In fact the life cycle GHG emissions for the biomass residue case are negative because of the decomposition that would have otherwise occurred, which would have resulted in methane emissions (see Figure 15). Additionally, Bain et al. also showed life cycle pollutant emissions from different power generation technologies (Bain et al., 2003) (see Figure 16).

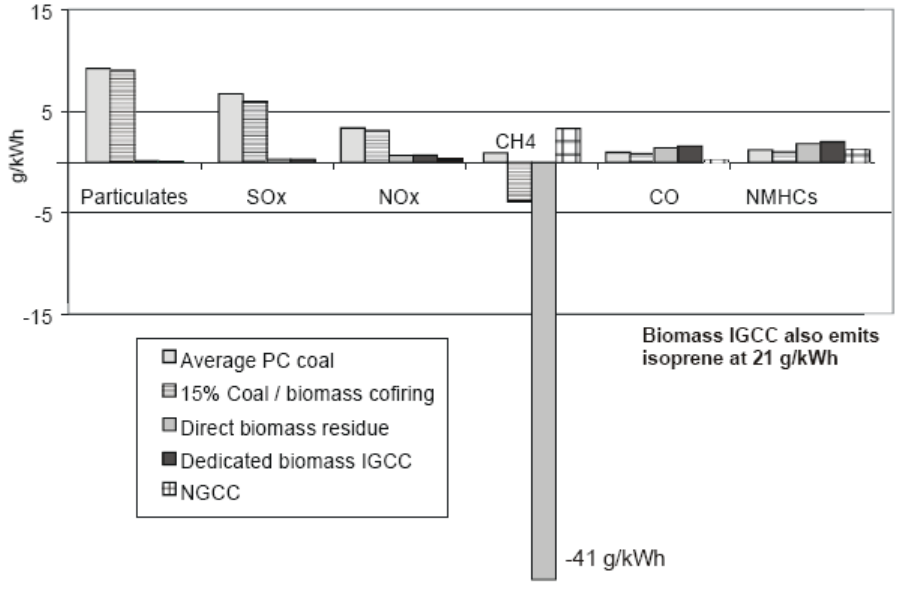


Figure 16: Life cycle pollutant emissions for several different scenarios of electricity generation (Bain et al., 2003)

To conclude, the implementation of dedicated energy crops must be considered carefully with the appropriate analysis of the life cycle environmental impacts resulting from those changes (land, water, albedo, soil health, etc.). The use of biomass waste and residue streams are more straightforward in their carbon reduction benefits and represent a lower risk path to increased use of renewable technologies than do the dedicated energy crops which if done incorrectly can have significant negative environmental impacts. Recall that Williams et al. showed the use of biomass waste and residue in California could contribute 11.9% of total electricity consumed in the state (Williams et al., 2007). Although the use of these waste and residue streams is more tractable in the near term, there is still risk of negative environmental impact particularly with regard to the pollutant emissions from these technologies as well as any additional GHG emissions that may occur due to changes in the transportation and processing of the particular waste/residue stream compared to normal operations.

3.1.3.2 Electricity Conversion Technologies

The environmental impacts associated with the electricity conversion technology itself are typically a large contribution to the pollutant emissions associated with biopower (Thornley et al., 2008). Waste and residue streams will also typically have lower emissions (GHG and pollutant) upstream of the electricity conversion technology, which emphasizes the importance of the environmental performance of the electricity conversion technology itself. Additionally, pollutant emissions occurring from biopower sources could have large air quality impacts if they are spatially located within urban air sheds with poor air quality; a significant concern in many regions of California.

From Figure 7, the largest potential for expansion of biomass residue utilization exists for forestry residues. The conversion technologies most applicable for use with this feedstock are gasification and direct combustion, as the use of anaerobic digestion would require the addition of water such that the solid content was reduced to less than 40% (Vandevivere et al., 2003). These conversion technologies are also applicable to those agricultural residues with high solid content (>40% solid content) and municipal solid waste. Direct combustion technologies exist commercially but exhibit low efficiencies and may have poor pollutant emission performance. Integrated gasification combined cycle systems will have higher efficiencies but remain in the development stages and are currently limited by high costs. Opportunities for modular, distributed small scale systems are also in development and make sense to the extent that biomass resources are diffuse and require collection and transportation to the point of conversion; whereas a modular system could reduce the need for this, and could potentially have cost benefits. However, if pollutant emissions from these distributed modular systems are high the potential for negative localized air quality impacts exists. The use of fuel cells with both small and large scale gasification systems could produce efficiency gains and reduce pollutant emissions although the efficiency gains could be highly beneficial to the small scale systems since fuel cell systems do not suffer from reduced efficiency at smaller scales like heat engines. Some typical numbers comparing the pollutant emission performance of gasification and direct combustion systems are shown in Figure 17 and Figure 18. Figure 17 and Figure 18 show differences in the emissions performance between the various gasification and combustion technologies despite the generality mentioned in earlier sections that gasification processes result

in cleaner plant operation (EPA, 2007). This further motivates the need to examine biopower installations on a case by case basis given that no general rules of thumb exist across the different thermal conversion technologies.

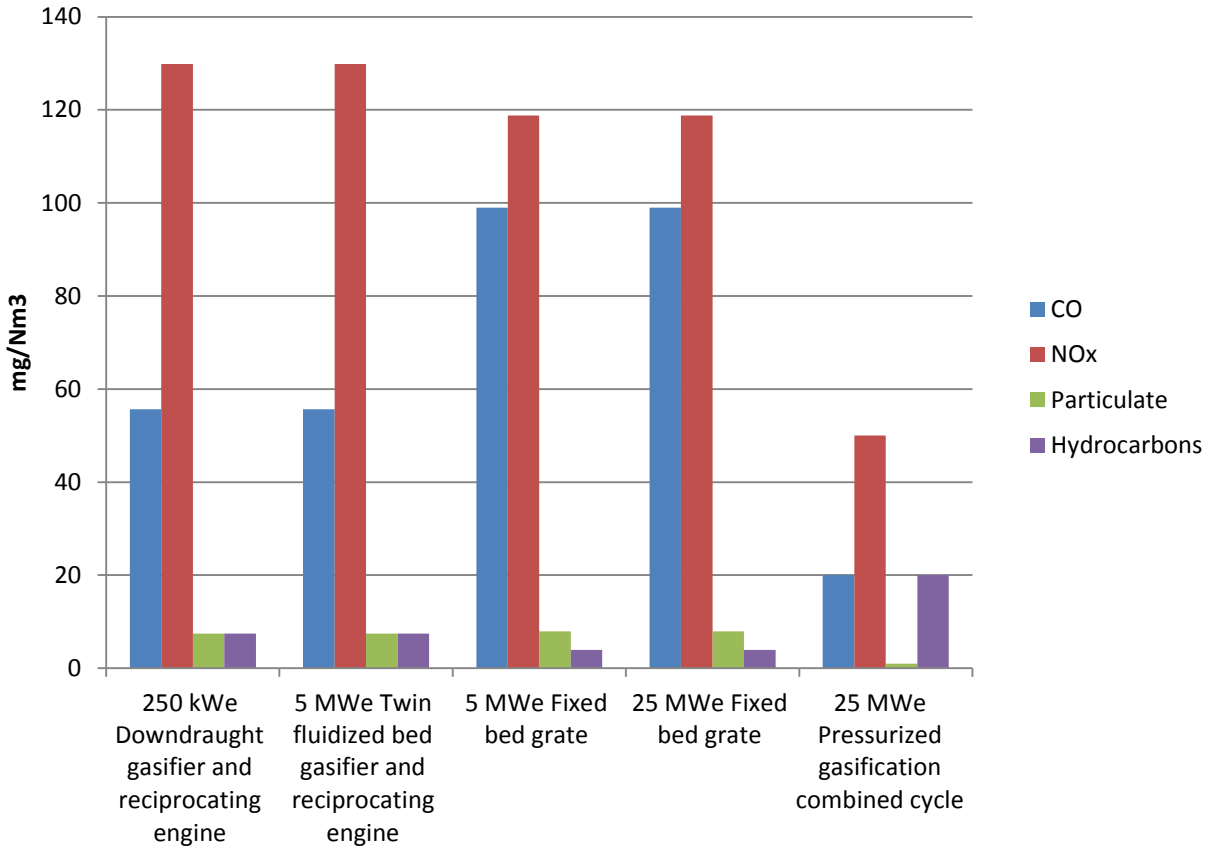


Figure 17: Emissions performance for several biopower technologies (Thornley, 2008)

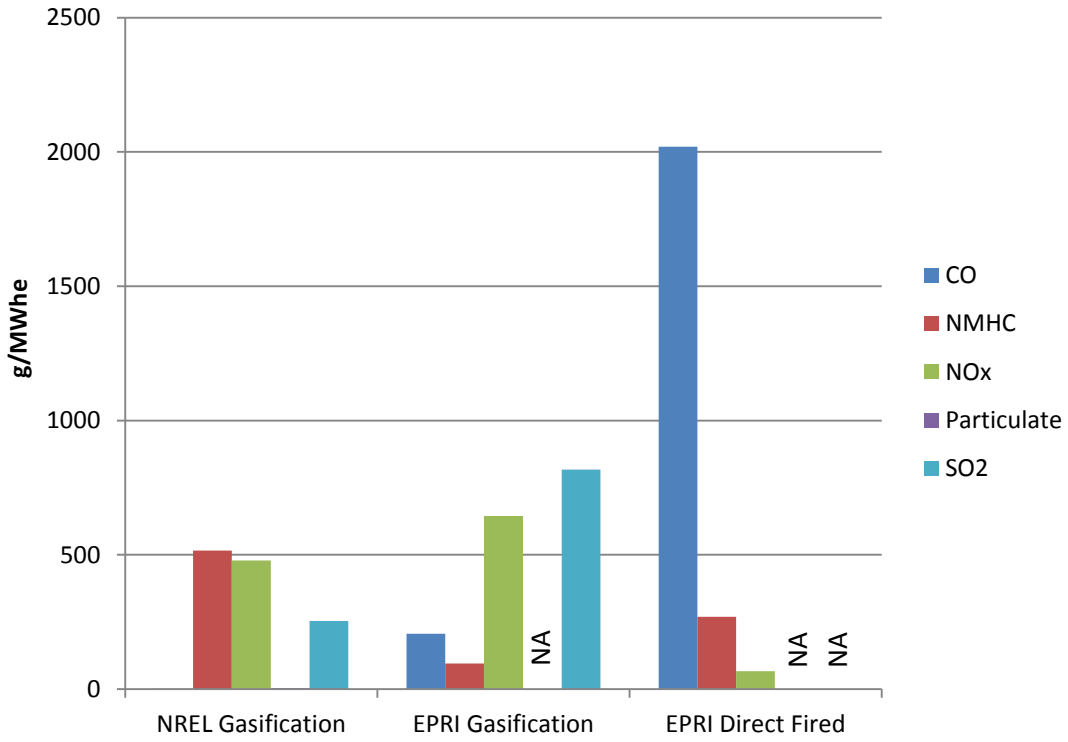


Figure 18: Emissions performance for several biopower technologies (Le et al., 2011)

The processing of agricultural residue and municipal solid waste for energy conversion can contribute to significant reductions in GHG emissions. In fact, simple changes in the management of municipal solid waste have led to significant reductions in GHG emissions from this sector (Weitz et al., 2002). These reductions are possible since any reduction in the emission of landfill gas has large GHG reduction benefits as a result of the methane content of this gas (50-80%) and the high global warming potential of methane, which is 28 times greater than CO_2. The implementation of gas collection systems at landfills for flaring or energy recovery has reduced GHG emissions by limiting these landfill gas emissions. However, these collection systems are not 100% efficient, and landfill gas is still emitted even in landfills with gas recovery (EPA, 1995). This issue and other issues related to land and water resources (leachate leakage) have led some countries to implement more sophisticated systems for management of MSW. These systems include high solid content anaerobic digesters, incineration facilities, gasification units, etc. (EC, 2001). Although the incineration or digestion of petroleum based products represents net GHG emissions to the atmosphere just as with fossil fuel fuels, the incineration or digestion of the organic (biogenic) fraction of municipal solid waste would lead to GHG emission reduction by eliminating the emission of landfill gas. Not all of the organic fraction of municipal solid waste should be handled in this manner because life cycle assessments have shown recycling to result in much larger GHG emission reductions than incineration (Finnveden et al., 2005; Moberg et al., 2005). Murphy et al. performed life cycle assessments of the GHG emissions associated with processing municipal solid waste using gasification, incineration, and anaerobic digestion using a commercial high dry solids content digester (DRANCO process by Organic Waste Systems) (Murphy et al., 2004). These researchers showed that use of the high solid content anaerobic digester provided the best GHG reductions when compared to the

scenario of flaring landfill gas. Finnveden et al. showed that the digestion of food waste provided the highest reductions in GHG emissions when compared to incineration and landfilling (Finnveden et al., 2005). This shows the potential of using these controlled anaerobic digesters for the processing of municipal solid waste. The European Commission also published a report in 2001 that analyzed the GHG emissions from several different waste management options. This report found that the anaerobic digestion of the organic fraction of municipal solid waste along with composting can lead to lower GHG emissions than the best practice landfill techniques that involve gas recovery for energy use and use of restoration layers (EC, 2001). The pollutant emissions associated with these processes as well as the electricity conversion of the biogas also remain an area of concern because in California these landfills may be located within non-attainment air basins and could then have significant effects on air quality.
Other 'wet' (low solid content) digester technologies are used to process wet waste such as manure and sewage. These wet digester technologies are currently utilized by waste water treatment plants and agricultural operations for processing animal manure with significant expansion possible in using animal manure for energy production (See Figure 7). These technologies provide GHG reductions as well and for similar reasons, i.e., decomposition leads to carbon emissions and use of the digester gas for electricity production can reduce these emissions. However, a similar problem remains: how do local emissions of pollutants from electricity conversion technologies (gas turbine, reciprocating engine, fuel cell, etc.) affect air quality.

3.1.4 Biopower Conclusions

Given that Executive Order S-06-06 requires 20% of the renewable electricity generated in California to come from biopower resources in 2010 through 2020 and in 2010 the biopower percentage of total renewable electricity generated was 17.5%, an increase in biopower capacity is expected in coming years. However, capacity increases could have negative environmental impacts, particularly with regards to localized air quality, for some generation pathways dependent on utilized feedstocks and conversion technologies. A major concern is pollutant emissions at the point of conversion, as well as emissions associated with the collection and transport of feedstock. The diffuse nature of waste/residue streams motivates the use of distributed biopower plants which could result in pollutant emissions in nonattainment regions (i.e. the San Joaquin Valley), however in centralized power generation situations the waste/residue streams require transportation, which also has associated pollutant emissions. Studies that assess potential air quality impacts across a range of different future year scenarios involving various deployment strategies of increased biopower capacity are needed. The spatial allocation of biomass resources performed by Williams et al. provides a starting point for such analyses considering that the spatial and temporal allocation of emissions sources is essential to air quality analyses (Williams et al., 2007). However, technically recoverable biomass resources may not be the actual recoverable resources due to economic or societal reasons; therefore, an assessment of the economically recoverable biomass resources under different scenarios would also be of worth. Important considerations in spatially and temporally resolved air quality impact studies include the many different conversion technologies available with currently limited available data (e.g., gasification technologies cannot be assumed to have a standard emission factors since these technologies have widely different emission factors depending on the design and manufacturer), therefore, it is important to use specific technologies that are applicable in the scenario under consideration.

In addition to the impacts on air quality, there are also issues related to water consumption and water/soil quality. Studies have shown that forestry residue removal in California mixed conifer forests does not affect the productivity of these forests, however, similar studies have not been completed for other types of woodlands and shrublands (Stewart et al., 2010). Water consumption in biopower plants will be similar to fossil fuel plants as both use similar thermodynamic cycles; although biopower plants utilizing fuel cell technology could have significant benefits for water consumption in that many fuel cell systems commercially available are water neutral. Water quality is an issue that is more difficult to address than water consumption and requires further analysis in conjunction with soil quality analyses.
Finally, the need to ensure that GHG reductions are actually achieved through the use of additional biopower resources is paramount due to the risk for other potential negative environmental impacts (i.e. local air quality disbenefits). For example, using municipal solid waste for the production of electricity may emit more GHGs than what recycling the material for re-use (e.g., paper), even if closed-vessel anaerobic digestion is utilized. GHG emission reductions throughout the life cycle of the feedstock-conversion technology pathway must be identified as not all pathways are equivalent in achieving reductions. Further, estimating emissions from biopower plants is essential in assessment of the effectiveness of California climate change targeted policy, such as programs related to AB 32.
It should also be noted that fuel cells and combined heat and power (CHP) systems can play an important role in addressing biopower related issues. Fuel cells can address two biopower related issues: air quality and water consumption. Fuel cells have very low pollutant emissions and can be sited in air basins with poor quality allowing distributed generation nearer to locations of waste/residue production. Most commercially available fuel cells designed for natural gas operation are water neutral. One challenge associated with fuel cells and biopower is their use with solid biomass typically burned or gasified. No commercial fuel cell units exist currently that will run with a syngas produced via a gasification process. However, with limited further development fuel cell systems could be adjusted for a syngas type fuel source. Combined heat and power can address the same two issues that fuel cells do but in a different manner; CHP increases system efficiency thereby reducing the amount of pollutants emitted and water consumed per unit of electric energy produced.

3.2 Biomass Derived Liquid Transportation Fuels

The use of liquid fuels produced from the conversion of biomass has gained considerable interest in recent years from both a GHG mitigation and energy independence stand point. Liquid transportation fuels that can be produced from biomass include ethanol and biobutanol produced from conversion of sugar, starch or cellulosic material, bio-diesel from oil crops such as soybean, and multiple fuels produced from the Fischer-Tropsch conversion process. Currently, ethanol produced from corn is the most widely used alternative transportation fuel in the U.S. with production levels of roughly 10.6 billion gallons in 2009, off-setting roughly 7 billion gallons of gasoline (RFA, 2010. Bio-diesel has the second highest production volume in the U.S., though significantly less than ethanol, at 491 million gallons in 2007 with 628 biodiesel refueling stations nationwide in 2009 (USDOT, 2010).

Third generation biofuels offer the potential for significant GHG benefits and include those produced from microalgae, including hydrogen, ethanol and bio-diesel. A benefit of algae-based

fuels is extremely high yields per acre, estimated to be a magnitude larger than conventional crops. Current biofuel yields are estimated at 50 gallons of biodiesel and 440 gallons of ethanol per acre for soybeans and corn respectively, while algae yields have been estimated at over 5,000 gallons per acre per year (Greene et al., 2011). Production of fuels from algae also avoids many of the issues concerning direct competition with food crops as algae growth does not require fertile land or high quality water. Algae growth may also offer a synergy with CCS technology as algae growth is accelerated by exposure to concentrated CO_2, such as from a power plant exhaust stream. However, strains of algae must be identified that have high oil content and resistance to viral infection. Further, costs associated with growing, harvesting, and fuel processing much be reduced. Due to these and other challenges, significant technological advancements in algae production processes are necessary prior to large scale commercialization, and it is unknown if high volumes of algae-based fuels will be available by 2050 (Wigmosta et al., 2011).

The GHG impact of biofuel use in the transportation sector is currently a source of significant scientific debate. A deep literature base of life cycle analyses displays wide ranging and contradictory values for quantified carbon intensities among different biofuels, and in some cases even for the same biofuel, depending on biomass feedstock, conversion technology, and life cycle energy requirements (Larson, 2006; Groode et al., 2008). Many factors influence whether the net environmental effects, including GHG and criteria emissions, are beneficial or detrimental (Börjesson, 2009). For biofuels to be viable GHG mitigation strategies GHG emissions must be reduced on a net life cycle basis relative to the displaced petroleum fuel. It is clear there is significant potential for mitigation as the uptake of carbon and soil carbon sequestration during growth of the biomass feedstock off sets much of the direct vehicle emissions occurring during fuel combustion. However, in parallel with direct vehicle emissions, upstream processes such as the agricultural practices associated with feedstock growth and harvesting (i.e. fertilizer and pesticide use, fossil fuel use in off-road farm equipment), transportation of feedstock, and bio-refining processes result in significant GHG emissions (Hill et al., 2006).

A factor that adds considerable complexity to estimating life cycle biofuel emissions is the impacts associated with direct and indirect land use changes (LUCs) (Escobar et al., 2009). Emissions from direct land use changes occur as a result of conversion of non-cropland (i.e. clearing of grassland or forest) into cropland to facilitate feedstock growth, releasing carbon sequestered in the soil. Emissions from indirect land use changes occur when cropland conversion occurs as a result of diversion of existing cropland elsewhere to facilitate biomass growth. Avoiding LUCs requires the continued increase in both the yields of biomass feedstock and the efficiencies of fuel conversion. Other factors include careful consideration of what areas are chosen for biomass plantations and responses by farmers to fluctuations in crop prices. Estimation of the magnitude of GHG emissions associated with land use change involves significant uncertainty and remains controversial, with some researchers arguing indirect LUCs actually result in negative life cycle GHG emissions relative to gasoline and others arguing biomass fuels can be produced without significant adverse LUCs (Searchinger et al., 2008; Tyner et al., 2010). As a result of this and uncertainties associated with other stages of fuel production, large variation is seen in the literature regarding life cycle GHG estimates for both ethanol and biodiesel.

The Federal Renewable Fuel Standards (RFS), adopted in 2005 and updated in 2007 as part of the Energy Independence and Security Act (EISA), establishes minimum volumes of renewable fuels to be used as a blend in on-road gasoline (Wiser et al., 2005). The most current version, RFS2, designates various sub-categories for renewable fuels and mandates life cycle GHG reduction thresholds for each category relative to conventional gasoline. In addition to conventional biofuel, the three added categories include non-cellulosic advanced biofuel, biomass-based diesel, and cellulosic biofuel requiring GHG reductions of 50%, 50% and 60% respectively relative to conventional petroleum fuels. The volumetric requirements federally mandated by 2022 are displayed in Figure 19. Bio-diesel is limited by feedstock availability and its application in the LDV sector is unlikely. As ethanol currently makes up the vast majority of the biofuel consumed in the U.S. today and is the only biofuel projected to expand significantly in the study period, particularly to meet RFS2 requirements, ethanol is the only biofuel pathways examined in-depth.

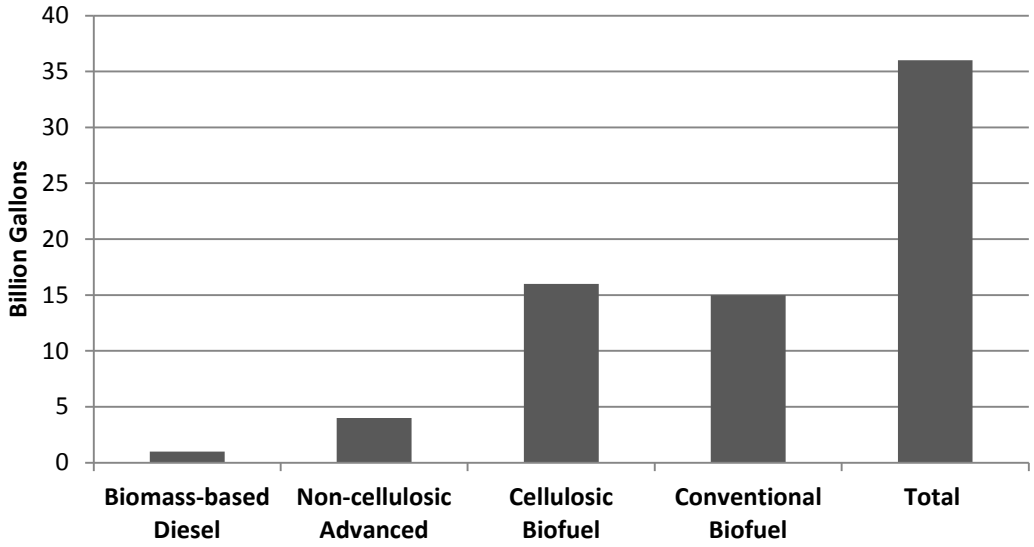


Figure 19: Federal RFS2 volume requirements mandated by 2022. Adapted from Greene, 2011

3.2.1 Ethanol

Ethanol can be produced from a variety of feedstock and production pathways. Current U.S. ethanol production relies heavily on corn as a cost effective, technically feasible, high-volume feedstock. Cellulosic materials that can serve as feedstock include switchgrass, prairie grasses, short rotation woody crops, agricultural residues, and forestry materials and residues. Cellulosic material compromise approximately 60-90% of terrestrial biomass by weight, allowing for a higher total percentage of feedstock utilization than corn, although breaking down cellulosic material into usable sugars requires additional processing. The increased complexity and processing times for cellulosic ethanol result in higher cost relative to corn-based ethanol although costs are expected to be reduced with increased commercialization (Greene et al., 2004). Future pathways for ethanol production that offer significant benefits from both a GHG

mitigation and sustainability perspective include production from algae, biomass waste, or from feedstocks farmed on abandoned agricultural land.

Ethanol has some intrinsic energy qualities such as a higher octane than gasoline which could have beneficial implications for efficiency and power in an internal combustion engine, particularly if the engine was optimized for ethanol (Brusstar et al., 2005). Research conducted by the NREL estimated that vehicle fuel efficiency increase for E10 and E85 vehicles could be up to 1-2 and 5.4% respectively (mile/BTU basis) (Tyson et al., 1993). However, the energy density of ethanol is roughly two-thirds that of gasoline, requiring a higher volume of fuel to be used for equivalent propulsion and necessitates the price of ethanol be two-thirds that of gasoline for economic competitiveness. An NRC committee concluded that for ethanol to be deployed economically, crude oil costs much reach 100 and 115 dollars per barrel gasoline equivalent (gge) for corn and cellulosic ethanol respectively (Figure 20) (NRC, 2009).

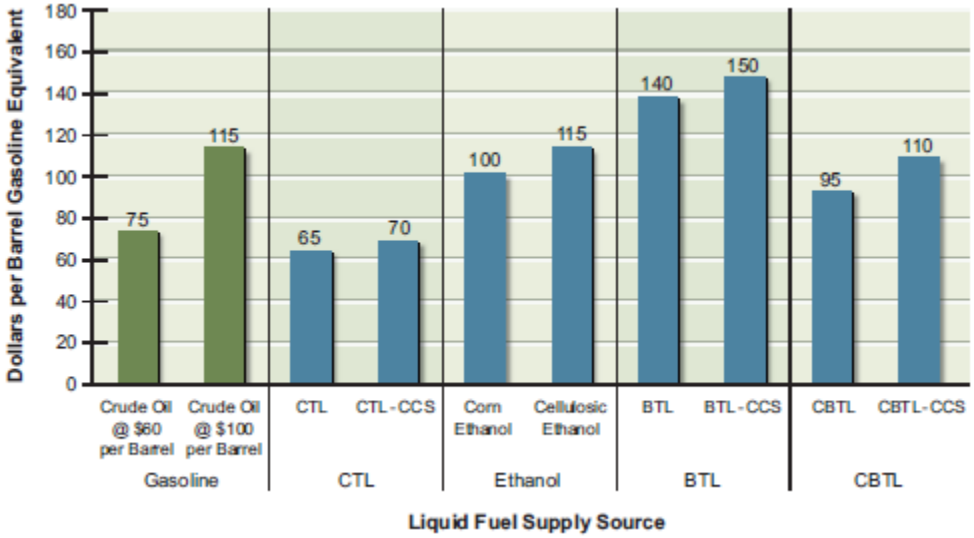


Figure 20: Estimated gasoline-equivalent costs of alternative liquid fuels in 2007 dollars. Note: BTL=biomass-to-liquid; CBTL=coal-and-biomass-to-liquid; CTL= coal-to-liquid fuel Source: NRC 2009[60]

By December 2014, 210 ethanol bio-refineries were in operation in the U.S. with an estimated capacity of 14.9 billion gallons annually and 3 new bio-refinery were under construction with a potential annual capacity of 100 million gallons (Renewable Fuels Association, 2014)[2]. The growing production trend is a result of such factors as the phase-out of methyl tertiary butyl ether (MTBE) and the Federal RFS2, which requires 36 billion gallons of biofuel, largely projected to be met with ethanol, be blended with gasoline by 2022 (U.S. CRS, 2011). Of this total, contribution of conventional biofuels – mostly corn ethanol – is capped at 15 billion gallons and 16 billion gallons must be cellulosic biofuels, having life cycle GHG emissions 60% below the 2005 average for petroleum fuel. Non-cellulosic advanced biofuel derived from renewable feedstocks which can be co-processed with petroleum is limited to 4 billion gallons and biomass-based diesel is limited to 1 billion gallons per year. These volumes are illustrated in Figure 19.

[2] Renewable Fuels Association, last update in December 2014. From: http://www.ethanolrfa.org/bio-refinery-locations/

Ethanol is blended with conventional gasoline in amounts per volume of up to 85% (E85) with E10 and E85 being the two most commonly available. Currently all light-duty vehicles in the U.S. can operate on E10 and ethanol flex-fuel vehicles (FFVs) can operate on E85, although currently FFVs have a small market share and are limited by lack of E85 fueling outlets (Andress et al., 2011).

A limiting factor for the availability of ethanol, and thus potential GHG mitigation, is the quantity of economically available biomass feedstock. Similar to the difficulty associated with emissions accounting, future volumetric feedstock estimates contain uncertainties regarding future crop yields, agricultural economics, national/state level policy, and others. An NRC committee concluded that in order to avoid increasing food prices only 25% of U.S. corn crops could be devoted to ethanol, limiting corn ethanol to about 12 billion gallons after 2015 (NRC, 2008). This would meet less than 6% of the reference case gasoline demand for LDVs for 2015 with the percentage falling in later years; however the assessment did not include sources for ethanol other than corn. A 2005 joint report by the U.S. DOE and the U.S. Department of Agriculture (USDA) estimated the future potential biomass resource available for energy production to be 1.3 billion dry tons per year from all sources, including starch, oil, and sugar food crops, energy crops such as rapid growth trees and grasses, agricultural residues, biomass wastes, and animal wastes (Perlack et al., 2005). The estimation assumed among others significant increases in crop yields, efficiencies for residue harvesting equipment, and improved land management strategies. Further, the estimations did not account for economic or resource allocation factors and should be taken as an upper bound as it is unlikely that all available biomass resources will be used for transportation fuel only. A joint study from Sandia National Laboratory and General Motors concluded that 90 billion gallons of ethanol annually could be feasible by 2030, but several conditions, including a minimum conversion yield of 74 gallons ethanol per dry ton biomass, were necessary (West et al., 2009). A study conducted by Andress, et al. (2011) accounted for competing demands for biomass resources, such as biopower plants, and capped the amount of available biomass in 2060 at 800 million dry tons annually, producing about 72 billion gallons of ethanol (Andress et al., 2011). *Reducing US Greenhouse Gas Emissions: How Much at What Cost?* reported in a mid-range case that production of biofuels could reach 30 billion gallons per year by 2030, equivalent to 14% of gasoline consumption, with 14 billion gallons derived from cellulosic biofuels (McKinsey, 2007). These studies demonstrate that though ethanol could be potentially available in the study horizon in considerable amounts, only a fraction of the liquid transportation fuel required to meet the huge projected demand in the LDV sector will be met. For example, in the extreme upper bound scenario in the DOE study, assuming an optimistic future conversion efficiency of 90 gallons ethanol per dry ton, the potential volume of produced ethanol would meet roughly 50% of the projected 2050 LDV transportation sector energy needs in the reference case developed by the NRC committee. Reported literature estimates of current and future feedstock availability and corresponding volumetric availabilities of ethanol are provided in Table 7.

Table 7: Current and future estimates of biomass feedstock and corresponding volumetric ethanol availability for use as a transportation fuel

Study	Year	Potentially Available Biomass [Tons]	Potentially Available Ethanol [gallons]	Potentially Available Ethanol [gge]
U.S. Production	2009	NA	10.6 Billion	7.067 Billion
U.S EPA RFS2	2022	NA	36 Billion	24 Billion (12% gasoline)
Perlack 2005	2030	1.3 Billion	137.4 Billion	91.6 Billion
McKinsey & Co.	2030		30 billion	20.1 Billion (14% gasoline)
NRC 2008	2015	25% U.S. Corn Crops	12 billion	8.04 Billion (<6% gasoline)
	2050	500-700 million cellulosic	45-63 billion	30-42 Billion (20% gasoline)

* Values in parenthesis represent the percentage of LDV fleet gasoline consumption displaced by the corresponding volume of ethanol

Estimates of the fuel carbon intensity of ethanol generally fall into two categories, estimates for corn ethanol and estimates for ethanol produced from cellulosic sources. Reported carbon intensities for corn ethanol vary significantly depending on assumptions regarding feedstock growth, production pathway, and LUCs. It has been argued that when LUCs are included in analyses of corn ethanol no benefits, and even net negative impacts, occur relative to petroleum fuels (Fargione et al., 2008; Hertel et al., 2010). Searchinger, et al. (2008) includes LUCs associated with conversion of forest and grassland to cropland and estimates that on a life cycle basis corn ethanol increases GHG emissions by 93% compared to gasoline. Hill, et al. (2009) estimates that when LUCs are included corn ethanol has no GHG benefits compared to gasoline if production occurs in a facility that uses natural gas for process heat, and GHG emissions increase by 28% if coal is used (Hill et al., 2009). However, other work has concluded that corn feedstock can be grown without large LUCs and improvements including crop yield increases and distillery efficiency mean corn ethanol can offer substantial life cycle GHG emissions reductions compared to gasoline (Greene et al., 2011). Tyner, et al. (2010) conducted a study involving comprehensive modeling of LUCs and concluded life cycle emissions of ethanol are 9.5-16.3% lower than those from gasoline (Tyner et al., 2010). Work by Wang, et al. (2011) estimates that current U.S. corn ethanol, on average, results in a life cycle reduction in GHG emissions of 24% compared to gasoline (Wang et al., 2011). The contrasting results from the Searchinger study was attributed by the authors to updated data reflecting technology improvements over time and detailed simulations in modeling LUCs. Another important factor in the carbon intensity of ethanol is the fuel source used to provide process heat and electricity to the ethanol plant. Wang, et al. (2007) examined different types of corn based ethanol plants and reports a full fuel LCA range of 3% increase to in GHG emissions if coal is used to generate necessary power to a 52% reduction if wood chips were used (Wang et al., 2007). It is clear that the carbon intensity of corn ethanol has experienced reductions as a result of technology improvements. Including LUCs, the EPA has concluded that corn ethanol produced in new, natural gas-fired production facilities will have emissions at minimum 20% below 2005 gasoline levels (U.S. EPA, 2010). Integrating biomass fuels such as wood chips or corn stover to produce heat and power further reduces the life cycle GHG emissions of corn ethanol. Kaliyan, et al. (2011) estimate reductions for corn ethanol compared to gasoline of 38.9%-119 % depending on the biomass conversion technology and system characteristics (Figure 21) (Kaliyan et al., 2011). Reductions over 100% without including carbon capture and sequestration indicate that the

production of biofuel co-produces electricity that is exported to the grid and displaces emissions from electricity generation from coal. The authors estimate that a reduction of 151.2% over motor gasoline would be possible for a biomass integrated gasification combined cycle (BIGCC) system utilizing corn stover as fuel in conjunction with sequestration of CO_2 in deep underground wells. Heath, et al. (2009) reported that E85 produced from corn-based ethanol in 2022 would offer a 40% reduction in global warming potential compared to 2005 gasoline, which is the standard set by the Federal EISA requirements (Heath et al., 2007).

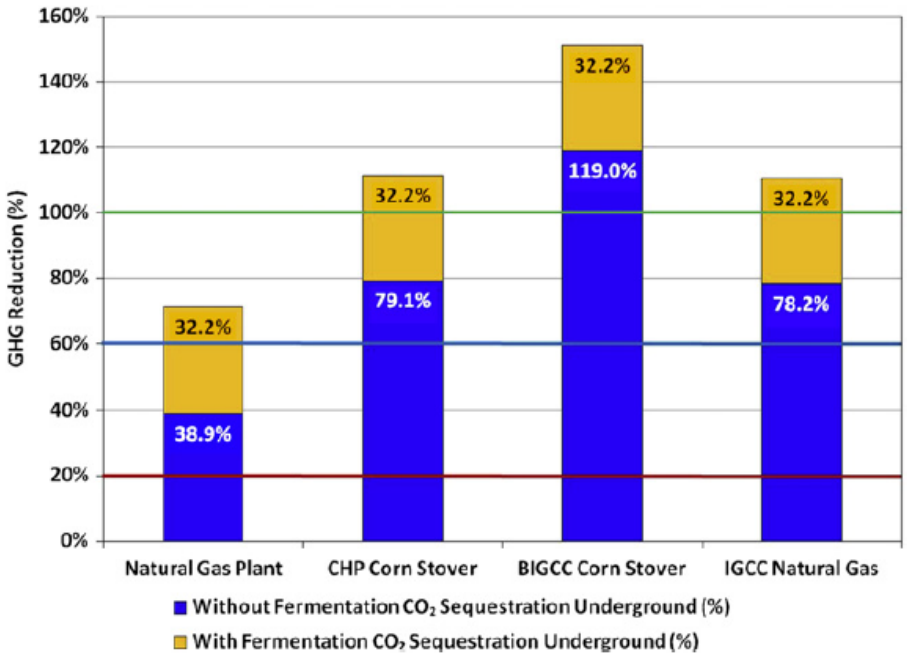


Figure 21: Percentage of lifecycle GHG reductions for corn ethanol compared to motor gasoline for plants utilizing various technologies and fuels. Source: Kaliyan et al., 2011

The most promising biofuel pathway, in terms of reducing carbon intensity, is ethanol produced from cellulosic biomass sources. The U.S. DOT estimates that life cycle GHGs for vehicles operating on E85 derived from ethanol produced from cellulosic sources is roughly half that of a vehicle operating on E85 produced from corn ethanol (USDOT, 2011). The consensus reached in the majority of studies is that ethanol produced from cellulosic feedstock does offer substantial reductions in carbon intensity relative to displaced petroleum fuels (one exception being the Searchinger study, which concluded that ethanol produced from switchgrass represented a 50% increase in emissions). Farrell, et al. (2006) estimated that ethanol produced from cellulosic sources could reduce GHG emissions by 90% with respect to gasoline (Farrell et al., 2006). Similarly a report issued by Argonne National Laboratory estimated that a vehicle operating on E85 produced from cellulosic sources would have net GHG emissions of 160 g/mile, equivalent to a 70% reduction relative to a baseline vehicle operating on gasoline (Brinkman et al., 2005). If improvements in cellulosic ethanol production allow for significant volumes of low carbon ethanol to be available GHG mitigation impacts could be significant. For instance, in the Sandia/GM study's reference case GHG reductions reached 400 $MMTCO_2eq$ per year in 2030, equivalent to offsetting emissions from 25% of the current fleet of gasoline vehicles.

Table 8: Estimates of LCA GHG Emissions for Various Ethanol Production Pathways with and without Estimates of Land Use Change Impacts. Source(s) CARB 2010 & Searchinger, et al. 2010

Study	CARB 2010 Without LUC [gCO_2eq/MJ] (LHV)	CARB 2010 With LUC [gCO_2eq/MJ] (LHV)	Searchinger, et al. 2010 With LUC [gCO_2eq/MJ] (LHV)
Gasoline	93.8	93.8	92
Corn-based Ethanol			177 (+93%)
Mid-West wet mill	75.1	89.8	
Mid-West dry mill, wet DGS	60.1	74.8	
Mid-West dry mill, wet DGS, 80% NG, 20% Biomass	56.8	78.3	
CA dry mill, dry DGS, NG	58.9	73.6	
CA dry mill, wet DGS, NG	50.7	65.4	
CA dry mill, wet DGS, 80% NG, 20% Biomass	47.4	62.1	
Sugarcane ethanol (Brazil)	27.4	73.4	
Cellulosic (Farmed)	5.4	23.4	138 (+50%)
Agriculture Waste	22.2	22.2	27 (-70%)

Extensive use of ethanol as a transportation fuel could impact criteria pollutant emissions spatially and temporally, leading to perturbations in ambient concentrations of air pollutants (Jacobson et al., 2007; Smith et al., 2009). Further, emissions of compounds labeled air toxics due to associated health effects may also increase. Direct vehicle emission perturbations from ethanol use are difficult to quantify as significant variation and contradictory values have been reported in the literature. Impacts on VOC and NO_x emissions are thought to be dependent on vehicle control technology and operating conditions, but the available data is somewhat unclear and a range of reported values exist in the literature (U.S. EPA, 2007a; Hsieh et al., 2002). It is known that adding ethanol in any capacity to gasoline increases the emissions of acetaldehyde (Gaffney et al., 2009; Knapp et al., 1998; Graham et al., 2008), but reduces others including benzene, a compound prevalent in motor gasoline (Yanowitz et al., 2009; Niven, 2005). In general studies have shown decreases in CO and total hydrocarbons in exhaust emissions for LDVs operating on E10 (USEPA, 2007a; Knapp et al., 1998; Poulopoulos et al., 2001), however others have shown equivalent or slightly increased emissions (Durbin et al., 2007). E10 use has been correlated with reductions in PM emissions relative to baseline gasoline, however PM increases substantially with decreases in temperature (Mulawa et al., 1997). With regards to NO_x, E10 use is generally correlated with increases in emissions (Hsieh et al., 2002; Reuter et al., 1992), although some studies have shown mixed results (Mulawa et al., 1997; He, 2003) and/or reduced emissions (Knapp et al., 1998). Graham, et al. (2008) conducted a statistical analysis of results from two studies as well as aggregate data reported in literature and reported statistically significant decreases in emissions of CO (16%), increases in emissions of NMHC (9%) and no statistically significant changes in NO_x, CO_2, CH_4 or N_2O. Ambient temperature was also important for NO_x emissions, for example vehicles operating on E10 at 75° F and 0° F showed decreased emissions but vehicles operating at -20° F showed increases in NO_x emissions relative to baseline gasoline (Knapp et al., 1998).

Criteria pollutant emission perturbations relative to gasoline differ for vehicles operating on E85 compared to E10. Graham, et al. reported statistically significant decreases in emissions of NO_x (45%), and NMHC (48%), statistically significant increases in acetaldehyde (2540%), and no statistically significant change in CO and CO_2 emissions. Similarly, a study examining emissions of Tier 1 and Tier 2 FFV operating on E85 reported reductions in NO_x of 54% and 28% and reductions in CO of 18% and 20% respectively (Yanowitz et al., 2009). E85 use has also been correlated with decreases in VOCs, which could have positive implications with regards to ozone formation. It is also important to consider associated increases in direct emissions of ethanol, which have been shown to be substantial and raise health and secondary air quality concerns. Further, in addition to tailpipe emissions, fuel evaporative losses have been shown to be 20-80% higher for E10 and E20 relative to baseline gasoline and are a major concern (Niven, 2005). A total emissions model of SoCAB, including evaporative losses, predicted lower CO emissions, equivalent NO_x, and higher acetaldehyde and ethanol emissions[92]. Differences across studies make accurate air quality impact assessment difficult, and can be attributed to such factors as fuel composition, test cycle, vehicle age, and emissions control technology.

Similar to evaluating GHG impacts, upstream emissions of pollutants, including those associated with feedstock growth, fuel production, and distribution, must be accounted for. Emissions associated with feedstock production occur from farm equipment, fertilizer and pesticide application, fugitive dust, and transportation of feedstock by rail, marine vessels, or trucks. Ethanol production facilities have significant emissions, as does the generation of energy that is consumed during the production process (Brady et al., 2007). Transport and distribution of ethanol and gasoline/ethanol blends via current shipping methods will result in increased emissions from trucks, ships, and rail unless a reliable pipeline infrastructure is developed (current gasoline pipelines can transport blends only up to 10% ethanol by volume). Transportation and distribution emissions include those associated with evaporative and spillage of fuel and could be important from an air quality perspective (Wakeley et al., 2009). A full LCA of criteria emissions for alternative/fuel vehicle systems demonstrated increase in total criteria pollutant emissions for E85 FFVs compared to gasoline vehicles, however reductions in urban emissions of up to 30% were reported due to the majority of emissions occurring from farming equipment, fertilizer manufacture, and ethanol plants, all of which are located in rural areas (Huo et al., 2009).

Detailed air quality modeling has demonstrated significant impacts on ambient air quality associated with fleet-wide ethanol use, particularly in regards to surface level ozone concentrations. Jacobson, et al. (2007) modeled the effects of 100% replacement of CVs with vehicles operating on E85 in Los Angeles and the U.S in the year 2020[78]. The study concluded that E85 use increased 24 hour and afternoon ozone up to 3 and 4 ppb respectively in L.A. and the Northeastern U.S., but decreased ozone concentrations in some areas of the Southeastern U.S. Further work by Jacobson, et al. (2008) compared air pollution health impacts from a conversion of on-road light- and heavy-duty gasoline powered vehicles to several alternative technologies including BEVs, HFCVs, and E85 and concluded replacement with E85 might increase the air pollution premature death rate by up to 185 deaths per year while significant health benefits were realized by BEV and HFCV replacement (Jacobson et al., 2008).

While a 100% fleet penetration of vehicles operating on E85 is not realistic in the 2050 horizon, these studies offer important insights into potential impacts and can be taken as upper bounds on potential impacts. On a regional scale Alhajeri, et al. (2011) compared regional photochemical pollution impacts in Texas from a 17% penetration of PHEVs to a 100% replacement with E85 and found that the highest reduction in maximum 1 hour ozone concentrations regardless of time of day occurred during PHEV scenarios (-8.5 ppb) and the maximum increase (2.8 ppb) occurred for the E85 scenario. An comprehensive EPA study examining the air quality impacts of the RFS2 mandated increase in ethanol consumption as a vehicle fuel concluded that ozone concentrations could increase by up to 1 ppb over much of the U.S., however several highly populated areas with poor ambient air quality experienced decreases in ozone concentrations. The observed improvements were likely a result of increased NO_x emissions in areas that are VOC-limited, which is not necessarily desirable. The study also demonstrated relatively small effects on air toxics other than increases in ethanol concentrations. Though the study was comprehensive the results are limited by uncertainties underlying data limits, for example $PM_{2.5}$ was not addressed due to an error in spatial emissions allocation that limited local-scale results.

4 Biomass Scenarios

4.1 Description of Biomass Scenarios

The list of scenarios analyzed in this report is designed to evaluate the potential impacts of biomass use for biopower using current technologies, and the potential effects of technological improvements for biopower production and of switching from biopower to biofuel production. The analysis is solely based on air pollutant and greenhouse gases emissions, and does not take economic parameters into consideration to determine the plausibility of the technology options. The list of scenarios is categorized in three major groups:

Group A: Increasing Capacity with Conventional Technology

These scenarios assume that the technology used for biomass/biogas conversion will stay the same as it is in existing installations. Solid residue facilities are typically solid-fuel boilers that power steam turbines to produce electricity and heat. Biogas installations are generally internal combustion engines, either reciprocating engines or gas turbines. This set of three scenarios assumes an increasing penetration of bioenergy installations assuming the existing mix of technologies. The end product of biomass conversion is the production of electricity and heat.

Biogas Installed Capacity:

1. Current biogas capacity:
    - Installed capacity of biogas-to-energy in the state is estimated to be ~ 370 MW from landfill gas, ~69 MW from digester gas from wastewater treatment plants, and nearly 4 MW from animal manure digester gas.[3]

[3] California Biomass Collaborative Bioenergy Facilities Database; http://biomass.ucdavis.edu/files/2013/09/11-20-2013-cbc-facilities-database_1May_2013_update.xlsx

2. Policy-driven new biopower from biogas:
    - SB1122 requires the CPUC to direct electrical corporations (IOUs) to procure 250 MW (cumulative, state wide) of new small biopower (less than 3 MW per project) in a separate IOU feed-in tariff program, of which 110 MW is for urban biogas and 90 MW for dairy and other agricultural bioenergy (that would include digester gas or small thermochemical conversion).
    - Governor Brown's Clean Energy Jobs Plan calls for 20 GW of new renewable generation by 2020: 8 GW would be large scale at 20MW or higher with 12 GW from distributed generation (presumes less than 20 MW per facility). Assume Gov.'s 20 GW goal is implemented with 20% met by biomass/biogas. Biogas facilities tend to be smaller than 20 MW and would be part of the distributed generation mix. Assuming that 20% of 12GW of distributed generation implies that 2.4 GW would be met by small scale new generation of biogas. However, this level of penetration is higher than the maximum potential for biogas, which is 1,130 MW. Consequently, biogas facilities are capped at the maximum potential levels.

3. Maximum potential for biogas based on current resources:
    - Potential biogas power capacity is approximately 175 MW from cow/cattle manure, 650 MW from landfill gas, 185 MW from food waste/green waste in current disposal stream and 120 MW from waste water treatment plants (does not include potential from food processing residues).[4] The total biogas capacity in this case is 1130 MW, which represents the maximum power capacity based on current biogas resources.

Solid-fuel Biomass Installed Capacity:

1. Current solid-fuel capacity:
    - There is approximately 725 MW of installed and operating solid-fuel bioenergy capacity in California (consuming forest, agricultural and urban residue).[5]

2. Policy-driven new biopower:
    - SB1122 requires the CPUC to direct electrical corporations (IOUs) to procure 250 MW (cumulative, state wide) of new small biopower (less than 3 MW per project) in a separate IOU feed-in tariff program, of which 50 MW are from material from sustainable forest management and 90 MW from agriculture (biogas or thermal conversion).
    - Governor Brown's Clean Energy Jobs Plan calls for 20 GW of new renewable generation by 2020: 8 GW would be large scale at 20MW or higher with 12 GW from distributed generation. Assume Gov.'s 20 GW goal is implemented with

[4] California Biomass Collaborative (unpublished) &
Williams, R. B., M. Gildart and B. M. Jenkins (2008). An Assessment of Biomass Resources in California, 2007. CEC PIER Contract 500-01-016, California Biomass Collaborative.
[5] CBC, Op. cit.

20% met by biomass/biogas. Assuming biomass facilities as part of the large scale mix (>20 MW), new biomass capacity would be 1.6 GW (20% of 8GW).

3. Maximum potential for solid-fuel (or thermal conversion):
    - Potential solid-fuel power generation capacity is approximately 620 MW from agricultural residues, 1910 MW from forestry resources and 1000 MW from the organic fraction of municipal solid waste.[6]

The overall installed capacity for both biogas and solid biomass installations is summarized in Figure 22. For the maximum potential case, the California Biomass Collaborative estimates overall potentials for urban, agricultural and forest waste, disaggregating the components of the "mixed" solid biomass category.

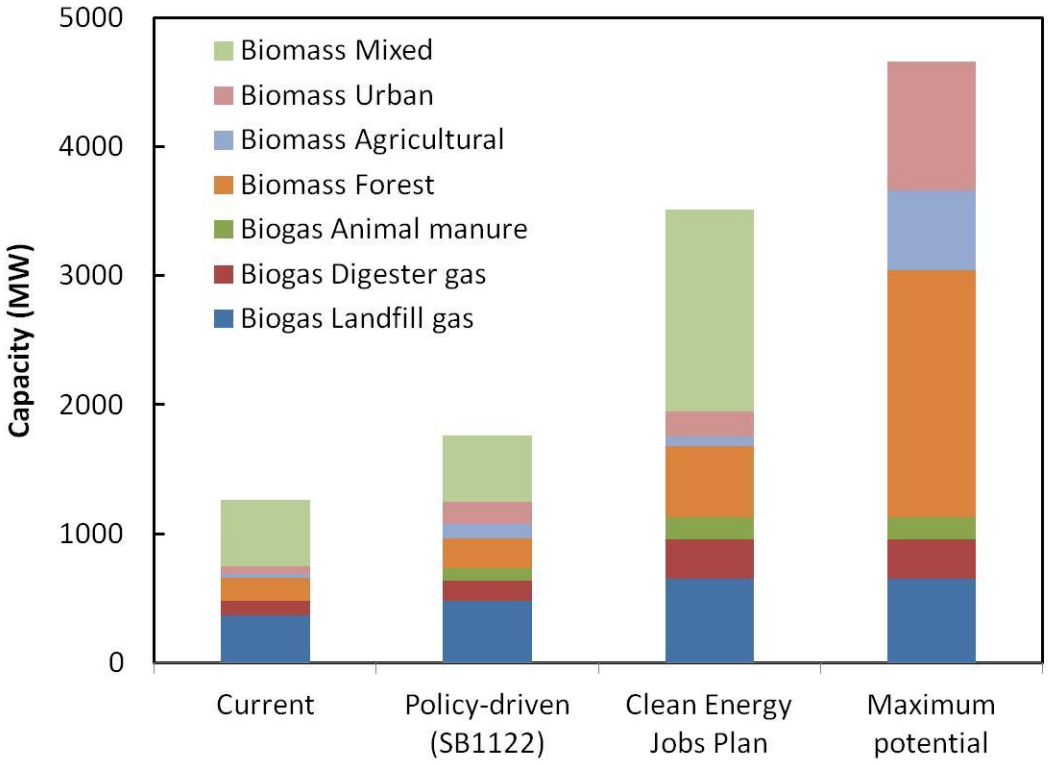


Figure 22: Summary of power generation capacity from biomass in scenarios with current biomass technology

[6] Ibid.

Group B: Technology Upgrade for Efficiency and Emissions

This group of scenarios assumes a shift in technology for both biogas and solid-fuel installations. For biogas installations, fuel cells will be used instead of internal combustion engines. For biomass installations, biomass-integrated-gasifier-combined-cycle is used instead of solid fuel boilers. The end products would still be electricity and heat. These technologies represent an improvement in emissions and total power production, due to lower emissions and improved efficiency. Maximum potential for both biogas and solid biomass is assumed.

Group C: Shift End Use from Electricity to Fuel

This group of scenarios assumes a shift in the end product from electricity and heat to renewable (and renewable synthetic) natural gas for vehicle fueling. Maximum potential for both biogas and solid biomass is assumed.

1. Production of compressed biomethane (a CNG like fuel) for vehicle fueling
   This scenario assumes that biogas will be cleaned and upgraded to biomethane, and compressed to be used for CNG vehicle fueling. Emissions from CNG vehicles will be added and emissions from gasoline/diesel vehicles will be displaced.
   Renewable-synthetic natural gas (RSNG) will be modeled from thermal conversion of solid biomass, and then compressed for fuel for CNG vehicles.

2. Production of pipeline quality biomethane for injection into natural gas pipeline
   This scenario assumes that biogas will be cleaned, upgraded and injected to the natural gas transmission and distribution system.
   Renewable-synthetic natural gas (RSNG) will be modeled from thermal conversion of solid biomass, and then injected to natural gas transmission and distribution system as well.

3. Assume co-digestion of bio-resources to produce (CNG)
   In this scenario, different streams of biomass will be co-digested to produce digester gas that will be cleaned-up and compressed to produce CNG for vehicles.

The yield in RSNG plants is calculated assuming a fraction of carbon mass in solid residue. Table 9 presents typical values for carbon content in selected residue types. For this study, the carbon content in grass is assumed as a conservative estimate for forest and agricultural waste. From the total carbon content in the residue, RSNG plants can achieve methane yields that range from 60% to 73% of maximum methane forming potential (Zwart et al. 2006). The range in yields depends on the configuration of the gasification process and the management of ashes formed, and for this study the lowest value is used to calculate RSNG potential.

Table 9: Carbon content of selected solid residues

Waste Type	Carbon content %
Forest residue	
Beech wood[1]	48.7%
Grass[1]	43.7%
Conifers[2]	50.0%
Angiosperms[2]	48.0%
MSW[3]	30.0%

[1]Zwart et al., 2006; [2]Thomas and Martin, 2012; [3]Bahor et al., 2008

An alternative to produce RSNG via gasification, solid residue can be treated to produce cellulosic ethanol as explained in Section 3. This bio-ethanol can be a substitute for the ethanol that is used for blending in gasoline. The theoretical yields of selected components of solid residue are presented in Table 10. For this study, the ethanol potential for agricultural waste is based only on the fraction of field and seed residue, because it is assumed that orchard and vegetable residues are not suitable for bio-ethanol production. Ethanol yield for forest thinnings is assumed to be representative of all forestry waste.

Table 10. Theoretical yields of selected components of solid residue[7]

Feedstock	Ethanol yields (gal/BDT)
Corn Grain	124.4
Corn Stover	113.0
Rice Straw	109.9
Cotton Gin Trash	56.8
Average yield	103.0
Forest Thinnings	81.5
Hardwood Sawdust	100.8
Bagasse	111.5
Mixed Paper	116.2
Switchgrass	96.7
Mixed feedstock	89.8

A second alternative to RSNG production for certain solid waste is the co-digestion of green and food waste in a high-solid anaerobic digester (HSAD). The high-solid digestate generates biogas similar to the one produced from wastewater treatment plants. The biogas can then be cleaned to produce CNG. A small fraction of the biogas is used for process heating. In addition

[7] Source: U.S. Department of Energy Bioenergy Technologies Office, Theoretical Ethanol Yield Calculator and Biomass Feedstock Composition and Property Database. http://www.afdc.energy.gov/fuels/ethanol_feedstocks.html

to biogas, 80% of the solid residue is converted into high-quality compost that can be marketed as soil amendment or fertilizer. Based on the ARB LCFS pathway for HSAD,[8] a plant would require 40/60 mix of green waste/food waste that would yield 2.29 MMBtu of biomethane per ton of residue. If the green waste and food waste fraction from MSW was used for HSAD, limiting the 40/60 mix ratio at county level, there is a potential for 4,858 BDT/day of residue that could yield 11,354 MMBtu/day biomethane. Similarly, if green waste from orchard and vine agricultural waste and waste from food industry was used for HSAD limiting the 40/60 mix ratio at county level, there is a potential for 5,421 BDT of residue that could yield 12,414 MMBtu/day of biomethane. Total potential for the production of biomethane from HSAD is 23,768 MMBtu, which is a small fraction of total potential for RSNG production.

Table 11 presents the maximum potential for biomethane production via RSNG from biogas and biomass resources in the state of California, and potential for cellulosic ethanol and biomethane from HSAD from solid residue. The total biomethane potential from biogas and biomass is more than $1.1 \cdot 10^6$ MMBtu/day. Assuming that CNG has an equivalency of 7.74 gallon of gasoline equivalent per MMBtu, this potential translates to approximately 8.9 million gallons of gasoline equivalent. Considering that projections from EMFAC suggest that gasoline consumption in 2020 will be 56.4 million gallons per day, CNG from biomass could potentially meet fuel demand of nearly 16% of gasoline vehicles in California. Conversely, taking into account that CA reformulated gasoline (CARFG) is a blend of 5.7%[9] ethanol and gasoline, bioethanol production from solid biomass could meet the entire state demand for ethanol blending for CARFG.

Table 11. Maximum potential for biomethane production from biogas and biomass, and potential for cellulosic ethanol production from solid biomass

		Biogas Potential (MMBtu/day)
Biogas	Landfill gas	177424
	Digester gas	83253
	Animal manure	47768
	Total	308445

		Biomass Potential (BDT/day)	RSNG Potential (MMBtu/day)	Ethanol Potential (gal/day)	HSAD CNG (MMBtu/day)
Biomass	Forest	30668	461110	2499430	
	Agricultural	10989	165231	382069	12414
	Urban	20679	213445	475769	11354
	Total	62336	839785	3357269	
Total			1148230		23768

[8] HSAD to CNG LCFS pathway: http://www.arb.ca.gov/fuels/lcfs/2a2b/internal/hsad-rng-rpt-062812.pdf
[9] California Energy Almanac: http://energyalmanac.ca.gov/gasoline/types_of_gasoline.html

Production of CNG requires a significant amount of power to clean-up biogas, generally using a pressurized filter, and to compress the biomethane at the required pressure for fueling or injection into pipeline. Based on ARB's LCFS pathways analysis, landfill gas purification requires 65,700 Btu of electricity per MMBtu of gas recovered.[10] In addition, assuming 98% efficiency in the compression stage recommended for the pathways for landfill gas to CNG and digester gas to CNG[11], the total electric power that would be required for RSNG clean-up and compression is 98,750 MMBtu/day. This is equivalent to 1,311 MW of new power generation, including grid losses of 8.1%.[12]

4.2 Emissions from Biomass Scenarios

As presented in Section 3, there are numerous ways of biomass utilization that can derive into a wide range in emission impacts. Even for the same type of technology, there exist a variety of emission factors that yields a range in the potential impacts of biomass use. We present here the emission factors of the most common technologies used currently for both solid biomass and biogas installations.

4.2.1 Conversion of Solid Biomass

Biopower production from solid residue in the state includes the following steps: collection and pre-processing of forest residue; transport to a biomass facility; and combustion in an average boiler. For urban and agricultural residue, its collection and transport to a disposal site occurs generally regardless of whether the residue is used for biopower or it is landfilled. Consequently, to calculate the air quality impacts of biopower from urban and agricultural residue it is assumed that no additional emissions from collection and transportation occur. In contrast, forest residue not used for biopower is generally left in the woods. Although some existing forest management measures may require the use of off-road equipment that results in pollutant emissions without using the residue for power, this study assumes that emissions from collection and transport of forest biomass should be accounted for.

Emissions from forest residue for electricity production are presented in Table 12. The calculations assume an average heat content of 9,000 BTU/lb for forest residue and emission factors for biomass boiler are based on the values used in CA-GREET 1.8b. Emissions from biomass collection are based on a comprehensive life-cycle assessment of biomass collection in California.[13] The lifecycle analysis included an estimate of fuel use, hours of operation and mass of forest residue collected and processed by over 20 different types of off-road equipment. Collection of forest biomass included both commercial thinning in plantations and industrial

[10] Landfill gas to CNG LCFS pathway: http://www.arb.ca.gov/fuels/lcfs/022709lcfs_lfg.pdf
[11] Low Carbon Fuel Standard pathways: http://www.arb.ca.gov/fuels/lcfs/workgroups/workgroups.htm#pathways
[12] Grid losses based on CA-GREET 1.8b
[13] LCA of Producing Electricity from CA Forest Wildfire Fuels Treatment, J. Cooper, 2008 - Included in Appendix 4 of Biomass to Energy: Forest Management for Wildfire Reduction, Energy Production and Other Benefits, CEC-500-2009-080-AP4. Emission factors based on EPA's NONROAD and MOBILE6 models

forest lands, and fire prevention operations in public lands. Emissions from transportation of biomass are based on NONROAD and MOBILE6 emission factors, and assume an average trip length of 60 miles from collection site to biomass plant. The biomass boiler emissions are based on the values used by CA-GREET 1.8b, which is in the range of emissions of biomass boilers inventoried by the California Biomass Collaborative.

In addition to direct emissions, Table 12 presents indirect emissions from the production of fuels required to operate the equipment to collect, pre-process and transport the forest residue. In total, collection and transport use 3.32 and 0.22 gallons of diesel per BDT of biomass, respectively. Emissions from diesel production are based on the values used by CA-GREET 1.8b.

Table 13 presents the contribution of the processes involved in the production of biopower from forest residue to the full lifecycle emissions. Overall, conversion of biomass to power is the biggest contributor to total emissions. More than 90% of NO_X, CO, PM and SO_X occur during combustion of biomass to produce power. Conversion also contributes to nearly 98% of total greenhouse gases emissions. Collection of biomass contributes to approximately 5% of criteria pollutant emissions, except for VOC, which contributes to nearly 14%, due to high VOC emissions from off-road equipment. Collection also contributes to nearly 2% of GHG emissions. The contribution of transport to total criteria pollutant emissions is less than 1% and its contribution to total GHG emissions is a small 0.01%. Finally, indirect emissions due to diesel production contribute to less than 2% in the emissions of NO_X, CO, and PM. Production of diesel contributes to 4% of total VOC emissions and 9% of total SO_X emissions, whereas its contribution to GHG emissions is less than 1%.

As shown in Table 13, the potential air quality impacts of biopower from solid residue depend in great part on the emissions from the conversion stage. Hence, any emission reductions in that stage will reduce the potential impacts of solid biomass use. As described in Section 3, combustion of solid biomass can be substituted with a gasification unit, which could potentially reduce emissions of air pollutants. Schueltze et al. (2010) conducted an analysis of several technological options for forest residue, and the overall performance characteristics are presented in Table 14. Just using current technology, switching from a direct fired boiler to an integrated gasification combustion unit, criteria pollutant emissions are reduced by an order of magnitude. In addition, next generation thermo-chemical conversion of solid biomass based on an integrated biofuels and energy production (IBEP) plant, NO_X and SO_X emissions from solid biopower from biomass could be further reduced. An additional benefit of using integrated gasification is an increase in efficiency in electricity production. Increasing power production from biomass will reduce the electricity needed from central power plants, hence potentially reducing emissions from the electric grid.

The IBEP plant (Shueltze et al., 2008) is an example of next generation biofuel production facility that integrates power and ethanol production. Other applications for biomass include the production of synthetic natural gas, which can then be used for heat and power generation, it can be compressed to produce CNG for vehicle or it can be used in the synthesis of Fischer-Tropsch fuels. There are numerous pilot plants and full scale operations in Europe and the United States.

[14] Because there is not available information on emissions from a synthesis gas installation, emissions for synthetic natural gas production are assumed to be similar to the emissions from the next-generation thermo-chemical bio-alcohol plant reported by Schueltze et al., (2010).

Table 12: Emissions from forest biomass use for biopower production

Process	**Harvest**	**Transport**	**Conversion**
Description	Biomass collection and pre-processing	On-road transport	Biomass Combustion
Equipment	Off-road equipment	Diesel Truck	CA average biomass boiler
Energy type	Diesel fuel	Diesel fuel	
Energy Use	3.32	0.22	
Energy Units	gal/BDT	gal/BDT	
Direct Emissions			
Units	lbs/BDT	lbs/BDT	lbs/BDT
VOC	0.0350	0.0011	0.2118
CO	0.1474	0.0010	3.0449
NO_X	0.2568	0.0044	4.3612
PM_{10}	0.0179	0.0020	0.5020
$PM_{2.5}$	0.0161	0.0018	0.2510
SO_X	0.0001	0.0000	0.1626
CH_4	0.0005	0.0000	0.1520
N_2O	0.0017	0.0000	0.4361
CO_2	68.2522	0.5032	3510.0
Indirect Emissions			
Units	lbs/BDT	lbs/BDT	
Description	Diesel production	Diesel production	
VOC	0.0093	0.0006	
CO	0.0255	0.0017	
NO_X	0.0730	0.0048	
PM_{10}	0.0089	0.0006	
$PM_{2.5}$	0.0041	0.0003	
SO_X	0.0149	0.0010	
CH_4	0.0956	0.0063	
N_2O	0.0002	0.0000	
CO_2	17.7808	1.1786	

[14] European Biofuels, Technology Platform: http://www.biofuelstp.eu/bio-sng.html

Table 13: Contribution (in %) to total emissions from processes in biopower production from forest residue

	Direct			Indirect
	Collection	**Transport**	**Conversion**	**Diesel**
VOC	13.59	0.43	82.15	3.83
CO	4.58	0.03	94.55	0.84
NO_X	5.46	0.09	92.79	1.66
PM_{10}	3.37	0.38	94.47	1.78
$PM_{2.5}$	5.89	0.67	91.83	1.61
SO_X	0.08	0.02	91.00	8.89
$CO_{2,eq}$	1.84	0.01	97.57	0.58

Table 14: Performance characteristics and emission factors for four different biomass energy plants (Schuetzle et al. 2010)

	Current Generation Biomass Combustion Power Plant	**Current Generation Integrated Gasification/ Combustion Power Plant**	**Next Generation Thermo-Chemical Conversion Power Plant**	**Next Generation Thermo-Chemical Conversion Bioalcohol & Power Plant**
Plant Size (BDT/day)	450	450	450	450
Electricity (kWh/BDT)	1000	1200	1400	550
Alcohol Fuel (gallons/BDT)	-	-	-	80
Diesel Fuel	-	-	-	50
Average Net Energy Efficiency	20%	22%	28%	50%
Emissions (lb/MMBtu output)				
NO_X	0.329	0.067	0.008	0.005
SO_X	0.125	0.010	0.002	0.001
PM	0.269	0.030	0.032	0.018
CO	0.897	0.070	0.042	0.023
VOC	0.085	0.018	0.003	0.002
CO_2	972	884	694	389

As described in Section 4.1, HSAD can be used for a fraction of MSW and agricultural waste that includes green and food waste. Table 14 presents the potential emissions per ton of residue from a HSAD plant that processes 100,000 tons of residues per year. Table 15 presents the emissions values per MMBtu of biomethane produced by the HSAD plant.

Table 15: Emissions from co-digestion of green and food waste in a high-solids anaerobic digestion facility with 100,000 tons per year processing capacity (emissions per ton of residue)

Process	**Handling/Processing**	**Plant Operation**	**Conversion**
Description	Biomass handling and compost processing	Electricity Use	Anaerobic Digestion
Equipment	Loader/Windrower	Waste handling and compression and purification of biogas	CA average biomass boiler for process heat
Energy type	Diesel fuel	Electricity	Biogas
Energy Use	0.09	0.22	0.05
Energy Units	MMBtu/BDT	MMBtu/BDT	MMBtu/BDT
Direct Emissions			
Units	lbs/BDT		lbs/BDT
VOC	0.0217		0.0002
CO	0.0813		0.0029
NO_X	0.1484		0.0030
PM_{10}	0.0088		0.0003
$PM_{2.5}$	0.0088		0.0003
SO_X	0.0016		0.0001
CH_4	0.0020		0.0001
N_2O	0.0002		0.0000
CO_2	15.5881		5.8720
Indirect Emissions			
Units	lbs/BDT	lbs/BDT	
Description	Diesel production	Electricity production	
VOC	0.0020	0.0109	
CO	0.0054	0.0270	
NO_X	0.0156	0.0345	
PM_{10}	0.0018	0.1364	
$PM_{2.5}$	0.0009	0.0353	
SO_X	0.0032	0.0093	
CH_4	0.0205	0.1355	
N_2O	0.0000	0.0013	
CO_2	3.8199	50.3084	

Table 16: Emissions from co-digestion of green and food waste in a high-solids anaerobic digestion facility with 100,000 tons per year processing capacity (emissions per MMBtu of biomethane produced)

Process	**Collection**	**Plant Operation**	**Conversion**
Description	Biomass collection and compost processing	Electricity Use	Anaerobic Digestion
Equipment	Loader/Windrower	Waste handling and compression and purification of biogas	CA average biomass boiler for process heat
Energy type	Diesel fuel	Electricity	Biogas
Energy Use	0.04	0.10	0.02
Energy Units	MMBtu/MMBtu	MMBtu/MMBtu	MMBtu/MMBtu
Direct Emissions			
Units	lbs/MMBtu		lbs/MMBtu
VOC	0.0095		0.0001
CO	0.0355		0.0013
NO_X	0.0647		0.0013
PM_{10}	0.0038		0.0001
$PM_{2.5}$	0.0038		0.0001
SO_X	0.0007		0.0000
CH_4	0.0009		0.0000
N_2O	0.0001		0.0000
CO_2	6.7991		2.5612
Indirect Emissions			
Units	lbs/MMBtu	lbs/MMBtu	
Description	Diesel production	Electricity production	
VOC	0.0009	0.0048	
CO	0.0024	0.0118	
NO_X	0.0068	0.0150	
PM_{10}	0.0008	0.0595	
$PM_{2.5}$	0.0004	0.0154	
SO_X	0.0014	0.0041	
CH_4	0.0089	0.0591	
N_2O	0.0000	0.0006	
CO_2	1.6661	21.9430	

4.2.2 Conversion of Biogas

Generation of biopower from biogas – landfill gas or digester gas – involves generally two steps: transmission from the point of biogas generation to the biopower plant, and combustion of the biogas in an engine, turbine or boiler. The transmission of biogas is accomplished with an electric blower that applies enough pressure to the biogas so that it can run through the cleanup

system (if any) and be fueled to the conversion device. Table 17 presents the emissions from biopower production from landfill gas using a Best Available Control Technology (BACT) engine.[15] The only direct emissions from this process occur in the combustion of biogas in the engine. Indirect emissions are accounted for the production of the electricity consumed by an electric blower. The emissions correspond to California marginal grid, obtained from CA-GREET 1.8b. The required power to transmit the biogas to the biopower plant is based on estimates by ARB, following the recommended low-carbon fuel standard pathway for CNG from landfill gas.[16]

Table 17: Emissions from landfill gas (LFG) use for biopower production

Process	**Harvest**	**Conversion**
Description	LFG recovery	LFG combustion
Equipment	Electric blower	BACT Engine
Energy type	Electricity	
Energy Use	9,262	
Energy Units	Btu/MMBtu	
Direct Emissions		
Units		lbs per MMBtu of gas recovered
VOC		0.2224
CO		0.6939
NO_X		0.1660
PM_{10}		0.0136
$PM_{2.5}$		0.0136
SO_X		0.0068
CH_4		1.1133
N_2O		0.0022
CO_2		143.6914
Indirect Emissions		
Units	lbs per MMBtu of gas recovered	
Description	Electricity for blower	
VOC	0.0003	
CO	0.0020	
NO_X	0.0033	
PM_{10}	0.0019	
$PM_{2.5}$	0.0006	
SO_X	0.0004	
CH_4	0.0045	
N_2O	0.0000	
CO_2	2.5496	

[15] Best available control technology (BACT) guidelines for a landfill gas engine in the South Coast Air Quality Management District, from: http://www.aqmd.gov/docs/default-source/bact/laer-bact-determinations/aqmd-laer-bact/ic-engine-a-n-391009-1850-hp.doc

[16] ARB LCFS pathway for CNG from Landfill gas: http://www.arb.ca.gov/fuels/lcfs/022709lcfs_lfg.pdf.

Table 18 presents the contribution of both direct and indirect sources of emissions to total emissions from biopower production from landfill gas. Except for PM_{10}, direct emissions contribute to more than 95% of total emissions of criteria pollutants. Indirect PM_{10} emissions are largely dominated by extraction of natural gas and petroleum products to produce the electricity in California. Finally, direct emissions of greenhouse gases comprise 98.5% of total emissions from biopower production from landfill gas.

Table 18: Contribution (in %) to total emissions from processes in biopower production from landfill gas

	Direct	**Indirect**
VOC	99.9	0.1
CO	99.7	0.3
NO_X	98.0	2.0
PM_{10}	87.8	12.2
$PM_{2.5}$	96.1	3.9
SO_X	95.1	4.9
$CO_{2,eq}$	98.5	1.5

Use of biogas from manure to produce biopower is similar to the process for landfill gas-to-energy applications. Assuming that the biogas is collected from a covered lagoon, the two main processes required for biopower generation from digester gas are compression using an electric blower, and combustion of biogas in an engine to produce power. Table 19 presents the emissions from biopower production with digester gas from dairy manure. The emissions assumed for the engine using digester gas are based on BACT guidelines,[17] and are comparable to the emissions from a landfill gas engine. Based on ARB estimates for a dairy biogas installation, the energy required for the electric blower is 22,209 Btu per MMBtu of recovered biogas.[18] Per unit of energy in the biogas, the required energy for the electric blower in a manure digester gas installation is more than twice the energy required in a landfill gas installation. As a result, the indirect emissions from digester gas recovery are more than twice as much as the emissions from collection of landfill gas. Table 20 presents the contribution of direct and indirect emissions from biopower production using digester gas. Because digester gas recovery is more energy intensive than landfill gas recovery, the contribution of indirect emissions from digester gas doubles the contribution of indirect emissions from landfill gas recovery for biopower production. For example, indirect emissions of NO_X add up to 4.4% of total emissions, and indirect emissions of PM_{10} correspond to 19.6% of total emissions. It is important to note, however, that a large fraction of indirect emissions from electricity use are

[17] Best available control technology (BACT) guidelines for a digester gas engine in the South Coast Air Quality Management District, from: http://www.aqmd.gov/docs/default-source/bact/laer-bact-determinations/aqmd-laer-bact/ic-engine-an-388050-1408-hp.doc)

[18] ARB LCFS pathway for CNG from dairy digester gas: http://www.arb.ca.gov/fuels/lcfs/022709lcfs_lfg.pdf. Electricity consumption to recover digester gas (11,124 Btu) + Energy to produce the electricity, including feedstocks (11,085 Btu)

related to the extraction of natural gas and other fuels required for electricity production. California imports over 90% of the natural gas it consumes,[19] and hence, most of the extraction of natural gas occurs outside of the state, thus having no effect on local air quality.

Table 19: Emissions from biopower production using biogas from manure

Process	**Harvest**	**Conversion**
Description	Digester gas collection	Biogas combustion
Equipment	Electric blower	BACT Engine
Energy type	Electricity	
Energy Use	22,209	
Energy Units	Btu/MMBtu	
Direct Emissions		
Units		lbs per MMBtu of gas recovered
VOC		0.2307
CO		0.7209
NO_X		0.1730
PM_{10}		0.0186
$PM_{2.5}$		0.0186
SO_X		0.0112
CH_4		1.1133
N_2O		0.0022
CO_2		143.6914
Indirect Emissions		
Units	lbs per MMBtu of gas recovered	
Description	Electricity for blower	
VOC	0.0007	
CO	0.0047	
NO_X	0.0080	
PM_{10}	0.0045	
$PM_{2.5}$	0.0013	
SO_X	0.0008	
CH_4	0.0108	
N_2O	0.0001	
CO_2	6.1136	

[19] Natural gas supply to California, Energy Almanac: http://energyalmanac.ca.gov/naturalgas/natural_gas_supply.html

Table 20: Contribution (in %) to total emissions from processes in biopower production from digester gas

	Direct	Indirect
VOC	99.7	0.3
CO	99.4	0.6
NO_X	95.6	4.4
PM_{10}	80.4	19.6
$PM_{2.5}$	93.3	6.7
SO_X	93.0	7.0
$CO_{2,eq}$	96.4	3.6

As in the case of solid biomass, emissions from biopower using biogas are dominated by the conversion stage. Reduction in the emissions from combustion of biogas in engines will reduce the overall impact of biopower on air quality. California Air Resources Board established emission standards for distributed generation facilities that limit the emissions from biogas generators substantially.[20] These limits are applicable for installations that are exempt from air district regulations, but the South Coast Air Quality Management District adopted the same restrictive limits. There are already several installations that use biogas to run microturbines to generate power and heat, and that have been certified by ARB to meet the restrictive air emission standards.[21] In addition to microturbines, biogas can be used in fuel cells, which emit at a lower rate than any other technology. In particular, emissions from fuel cells are 2 orders of magnitude lower than a biogas engine. Hence, the use of fuel cells to produce power from biogas would significantly reduce the emissions from biopower production. Table 21 presents a comparison of emissions between an engine and a fuel cell.

Table 21: Performance and emissions comparison between a biogas engine and a fuel cell

	Engine	Fuel Cell[22]	ARB limits
Efficiency	0.34	0.47	
Emissions (lb/MWh)			
VOC	2.23	--	0.02
CO	6.96	--	0.10
NO_X	1.67	0.01	0.07
SO_2	0.07	0.0001	
PM_{10}	0.14	0.00002	
CO_2	1441	940	

[20] DG emission regulations: http://www.arb.ca.gov/energy/dg/2006regulation.pdf
[21] http://www.arb.ca.gov/energy/dg/eo/eo-current.htm
[22] http://www.fuelcellenergy.com/why-fuelcell-energy/benefits/ultra-clean/

4.2.3 Emissions Displacement from Biomass Use

The assessment of the impacts of biomass needs to account for any displacement of emissions that the use of biomass may provide. For example, new biopower production from biomass will displace power generation that otherwise would have been produced by the existing California grid. New fuel production from biomass, whether it is CNG or ethanol, will displace fuel production and consumption that would otherwise been produced by the current infrastructure of oil refineries in the state. For CNG vehicles, in addition to the emissions displaced from gasoline and diesel marketing, emissions changes due to the shift from gasoline/diesel to CNG engines must also be accounted for. It is not clear however, whether a decrease in gasoline and diesel demand would translate into a decrease in petroleum refining. For this study, we assume that even though CNG or ethanol from biomass could displace a significant portion of the fuels consumed in the state, petroleum refining will remain unaffected as the excess in production could be exported to other parts of the US. However, emissions from petroleum marketing which involves transporting fuel to fueling stations would be affected if gasoline and/or diesel is displaced significantly by CNG.

4.2.4 Summary of Emissions from Biomass Scenarios

The analysis of the emissions from all scenarios includes four major contributors to total emissions from biomass use: (1) feedstocks, (2) collection and transport, (2) conversion and (4) savings.

(1) Feedstocks: emissions from feedstocks refer to all the emissions relates to all indirect emissions that occur during the production of electricity and fuels that are used to operate machinery and processing plants for biomass collection, processing and conversion. Sources of feedstock emissions include: emissions from diesel production for fueling off-road equipment that collects forest residue and loads residue in processing plant, and emissions from electricity production required to power biogas blower, processing plant electrical needs and biomethane compressor.

(2) Collection and transport: emission from collection and transport is only considered for the collection of forest residue. This study assumes that any other solid residue, e.g. MSW and agricultural reside, is collected regardless of whether the residue is used for biopower production. As a result, production of power or fuels from residues other than forestry waste does not incur in additional collection and transportation emissions, and hence, no emissions from this stage are accounted for.

(3) Conversion: emissions from conversion include all direct emissions that occur in the biomass processing plant. Conversion processes include: combustion of biomass or biogas in biopower production, partial oxidation of biogas in the biogas clean-up process, and gasification of biomass for the production of synthesis natural gas.

(4) Savings: emission savings include all the emissions displaced by the production of power and fuels from biomass. When biogas and biomass are used to produce biopower,

emissions from the production of the same amount of power using California's grid should be subtracted. Similarly, when biogas and biomass are used to produce pipeline-grade natural gas, emissions from the production of California natural gas should be subtracted. In the specific case that biomass is used to produce CNG to fuel gasoline vehicles, emissions from the production of equivalent gasoline fuel need to be subtracted. In addition, emissions from switching from conventional gasoline vehicles to CNG vehicles need to be accounted for.

The analysis is focused on the emissions of NO_X, PM and greenhouse gases expressed as emissions of CO_2 equivalent. NO_X and PM are the most relevant criteria pollutant for the formation of ozone and particulate matter in California. Emissions of CO_2 equivalent include contribution of CH_4 and N_2O, which are emitted at much lower rates than CO_2, but because their global warming potential is 34 and 298 times CO_2 warming potential,[23] respectively, they can contribute sensibly to total climate forcing. A fraction of PM emissions is formed by black carbon (BC), which is known to be a short-lived climate forcing compound. BC contributes to global warming, but it has a relative short atmospheric lifetime. This implies that reduction of BC emissions could dissipate their global warming effect rather quickly, compared to long-lived compounds like CO_2.

Figure 23 presents the emissions for all scenarios in group A: Increasing Capacity with Conventional Technology. All these cases assume that both biogas and biomass are used to produce power by using a biogas engine and a biomass boiler. Emissions are disaggregated between biogas and solid biomass applications. As described in Section 4.2.1 and 4.2.2, emissions from conversion dominate the overall emissions from biopower production. There are no emissions associated to biomass collection and transport in biogas applications, other than the electricity required for the blower to pump the landfill gas and the digester gas to the biopower facility. For biomass, emissions from collection and transport of only forest residue are accounted for.

Emissions of NO_X from current facilities are approximately 45 tons/day, and increase to up to 157 tons/day in the case of maximum potential for biopower production. According to ARB, [24] total statewide emissions for 2012 are 2,162 tons/day, and are expected to decrease to 1,610 tons/day by 2020. This implies that emissions from current biopower plants contribute to 2.1% of total statewide NO_X emissions. In addition, assuming that the maximum potential could be achieved by 2020 using current technology, potentially biopower would contribute to 10% of total statewide NO_X emissions by 2020.

Emissions of PM from current facilities are approximately 5 tons/day, and increase to up to 17 tons/day in the case of maximum potential. ARB estimates for statewide PM are 1,963 tons/day in 2012 and 1,921 tons/day in 2020. Hence, the contribution from biopower could grow from 0.3% with current facilities to 0.9% in 2020 with maximum potential for biopower production using current technology. The impact of biopower on primary PM is less pronounced than the

[23] Global Warming Potential values from the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, 2013. Values include climate-carbon feedbacks, and present an increase in the CH4 GWP from 25 to 34.

[24] ARB Emissions Inventory Data: http://www.arb.ca.gov/ei/emissiondata.htm

effect on NO_X emissions. However, it is important to note that NOX can participate in the formation of secondary PM. Consequently, to account for the overall effect of biomass use on PM concentrations in the state, air quality simulations are required to quantify the formation of secondary PM in addition to the contribution from direct PM emissions.
Emissions of CO_2 equivalent are approximately 37,000 tons/day and could increase up to 151,700 tons/day in the maximum potential case. ARB's estimates for statewide GHG emissions are 460 million tons of $CO_{2,eq}$ per year in 2012 (1.2 million tons/day),[25] and projected to grow up to 600 million tons/year in 2020 (1.64 million tons/day), in a business-as-usual projection. [26] With these GHG emission estimates, biopower production contributes to nearly 3% in total in-state $CO_{2,eq}$ emissions currently, and could increase to 9.2% in 2020.

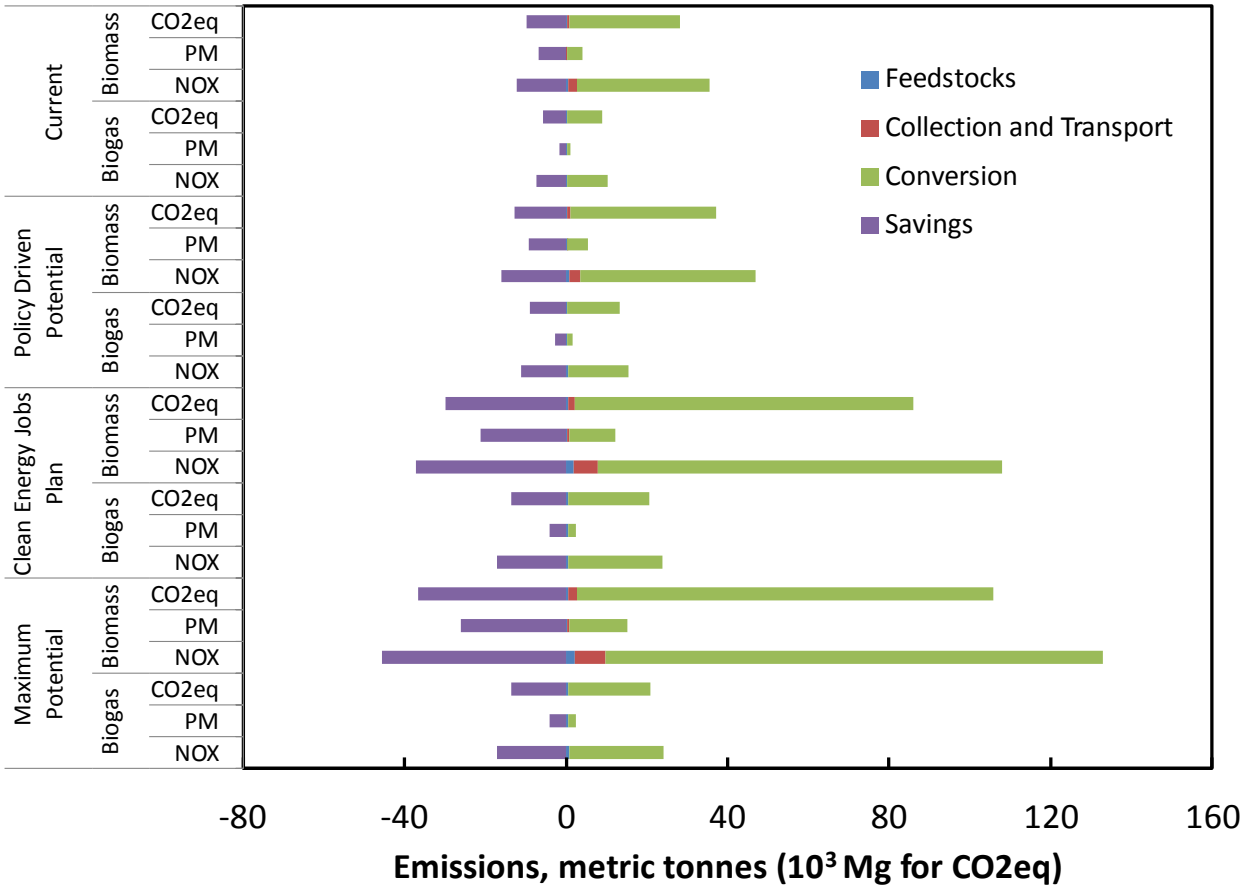


Figure 23: Summary of emissions from biomass in scenarios with current biomass technology (group A)

In addition to direct emissions, Figure 23 shows the potential savings in emissions due to displacing emissions from power generation by biopower production. Figure 24 shows the net emissions for the scenarios in Group A. For NO_X and $CO_{2,eq}$, savings do not totally offset emissions from biopower production. Namely, emissions from biopower using current

[25] California Greenhouse Gas Emission Inventory, http://www.arb.ca.gov/cc/inventory/inventory.htm
[26] California 1990 Greenhouse Gas Emissions Level and 2020 Limit, http://www.arb.ca.gov/cc/inventory/1990level/1990level.htm

technology are higher than the sum of direct and indirect emissions generated from producing the same amount of electric power by the existing grid, and the net emissions presented in Figure 24 are positive. On the contrary, savings in PM for both biogas and biomass applications are larger than direct emissions, and as a result, net emissions for the entire fuel cycle are negative. However, it is important to note that for NO_X and PM, some of the emission savings occur out of state. Emission savings include emissions from the extraction of natural gas and other fuels in other parts of the country and the world that are required for power generation. Based on the emissions shares by CA-GREET 2.0,[27] using California current mix for in-state power generation and assuming that approximately 33% of the power is imported,[28] the portion of emission savings that occur in the state is shown in Table 22.

Table 22: Fraction of the emissions savings for biopower production for selected pollutants that occur in the state.

Pollutant	Fraction of in-state Savings
NO_X	37.8%
PM	24.9%
$CO_{2,eq}$	61.8%

As a result, those savings in criteria pollutant emissions do not have a direct effect on regional air quality in the state. It is also important to note that savings in GHG emissions do not include emission credits for the use of biomass. For example, forest residue not used for biopower may be disposed of by prescribed burning, or left to decompose in the forest. Biogas not used for biopower could either be vented or flared. Hence, not using biomass for biopower can result in emissions of GHG that are not included in the emission savings. Including these GHG emission credits would reduce the carbon footprint of biopower production, and thus the results shown in this section represent an upper bound for GHG impacts.

[27] CA-GREET 2.0 available at: http://www.arb.ca.gov/fuels/lcfs/ca-greet/ca-greet.htm
[28] California current mix for in-state power generation and imports from: http://energyalmanac.ca.gov/electricity/total_system_power.html

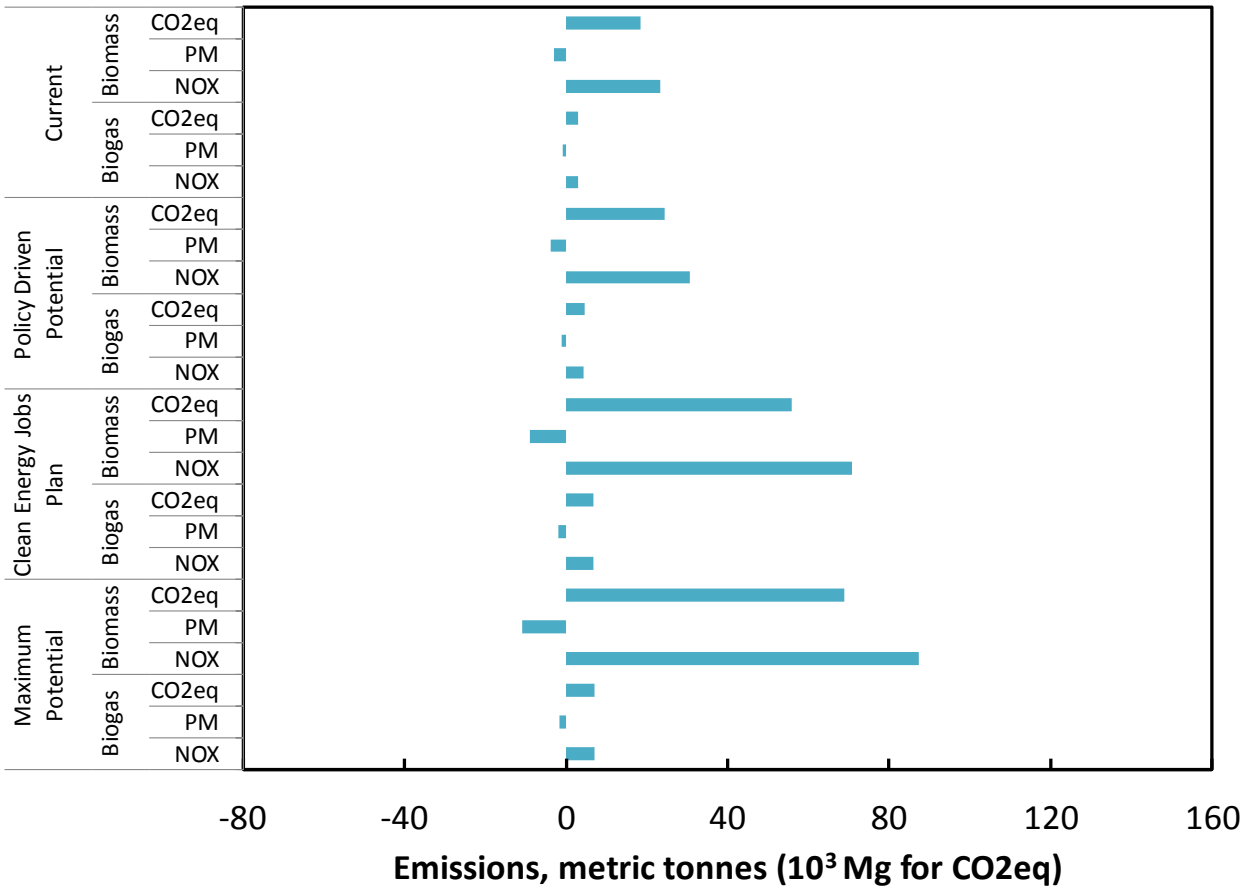


Figure 24: Net emissions from biomass in scenarios with current biomass technology (group A)

Figure 25 presents the emissions from Group B: Technology Upgrade for Efficiency and Emissions, in comparison with the case with maximum potential for biopower with current technology. Technology upgrades consist of switching current boilers and engines with next generation gasification systems and fuel cells. The result is a significant decrease in direct emissions of criteria pollutants with respect to the case with current technology. Direct GHG emissions do not change, as the same amount of carbon is converted into CO_2, but because of the increase in efficiency in power generation, emission savings are also increased with respect to the case with maximum potential and current technology.

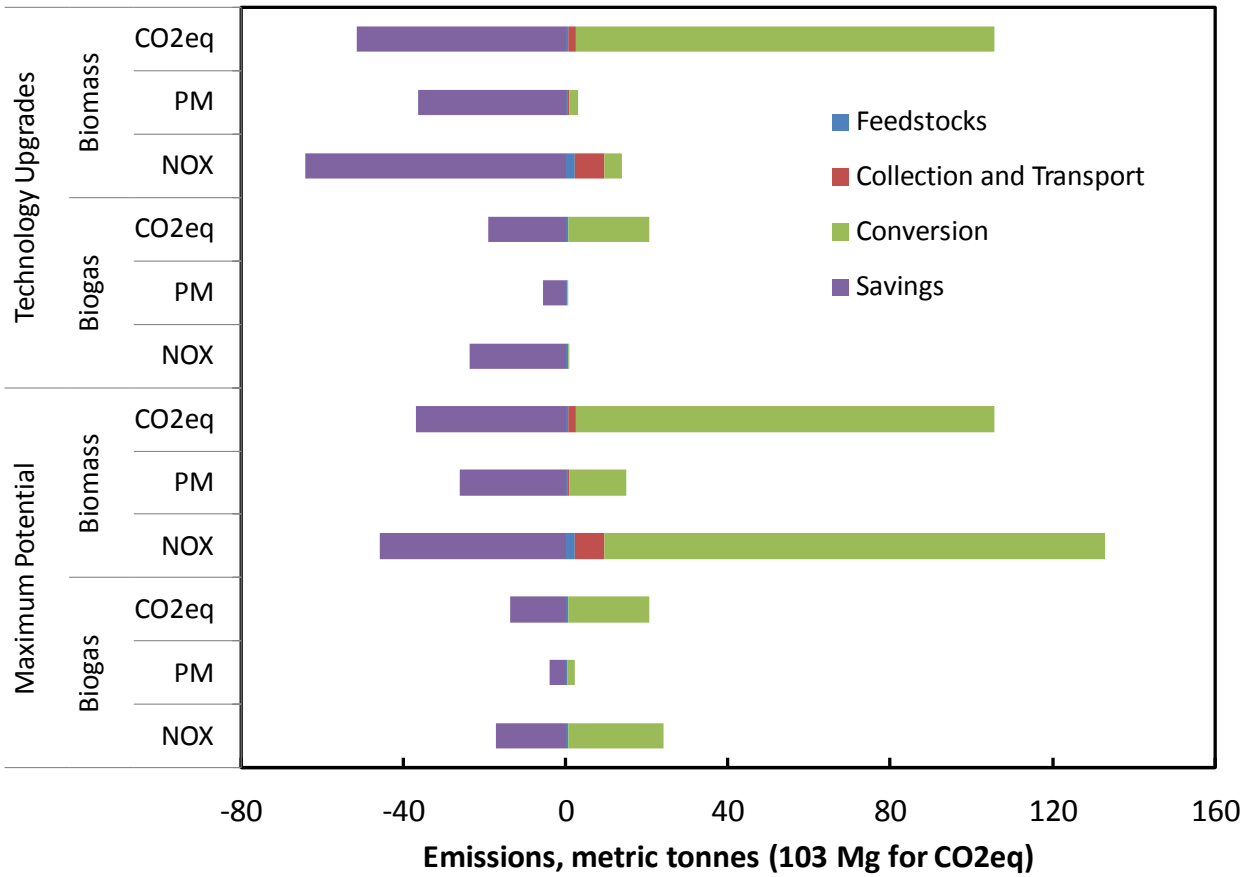


Figure 25: Comparison of emissions from biomass in scenarios with maximum biomass potential with current technology (group A) and with technology upgrades for efficiency and emissions (group B)

Resulting net emissions from group B are presented in Figure 26 together with net emissions for the maximum potential with current technology. Because of the very low emissions from fuel cells and integrated gasification systems, net emissions of NO_X and PM are negative for the entire fuel cycle. As stated above, it is important to note that a large part of the savings in criteria pollutant emissions occur outside of the state (as shown in Table 22), having no effect on air quality. Regarding GHG emissions, technology upgrades decrease net emissions of CO_{2eq} by 26% with respect to the current technology case.

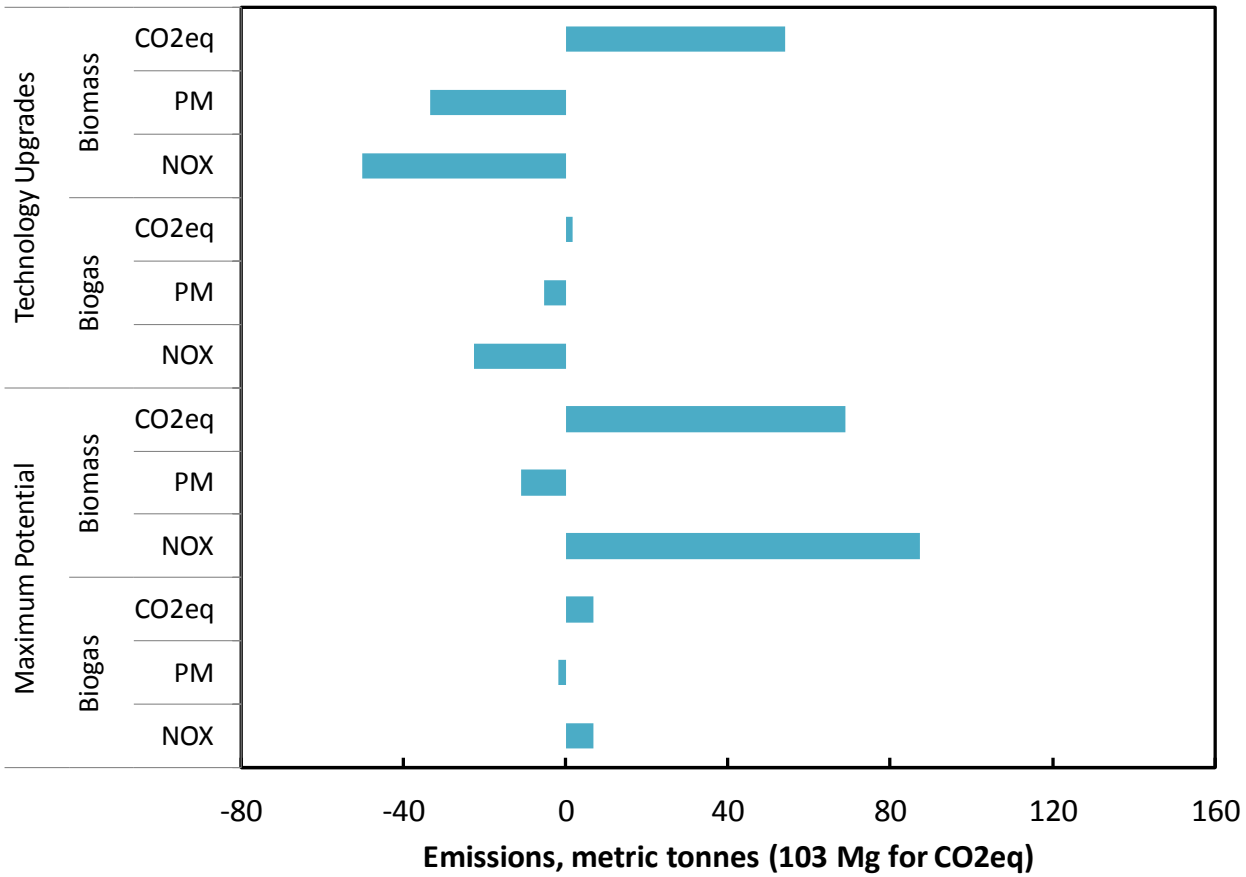


Figure 26: Net emissions from biomass in scenarios with maximum biomass potential with current technology (group A) and with technology upgrades for efficiency and emissions (group B)

Figure 27 presents the emissions of scenarios in Group C: Shift End Use from Electricity to Fuel, together with the case with maximum potential for biopower with current technology. Group C includes two cases with generation of CNG from solid biomass via gasification: one dedicated to produce CNG for vehicle consumption and the other one for pipeline injection. Direct emissions from these two cases are the same, because the processes to generate the CNG are the same in both cases. Emissions from feedstocks in these two cases are considerably higher than in the cases of group A and B, because more energy is required to clean-up biogas and synthesis gas, and to compress them. The only difference between these two CNG scenarios is the emissions displaced by the CNG. In the case that CNG is dedicated to vehicle consumption, emission displacement is due to the savings in gasoline production and marketing needs that production of CNG from biomass provides. In addition, the case includes savings in emissions from vehicles switching from gasoline to CNG consumption. Conversely, in the case that CNG is dedicated to pipeline injection, emission displacement is calculated from the savings in natural gas production and marketing demand that CNG provides. No additional savings are considered in this CNG case as combustion of NG from biomass is assumed to produce the same pollutant emissions as combustion of conventional NG. Hence, comparing the two cases is analogous to contrasting

emissions from equivalent energy units of gasoline and natural gas. The result is that producing gasoline for California is more pollutant-intensive than producing natural gas, and thus, reducing gasoline production achieves higher emission savings than reducing production of natural gas containing the same amount of energy. Consequently, on a full fuel cycle emissions standpoint, producing CNG for vehicles is more favorable than producing natural gas for pipeline injection as shown in Figure 28.

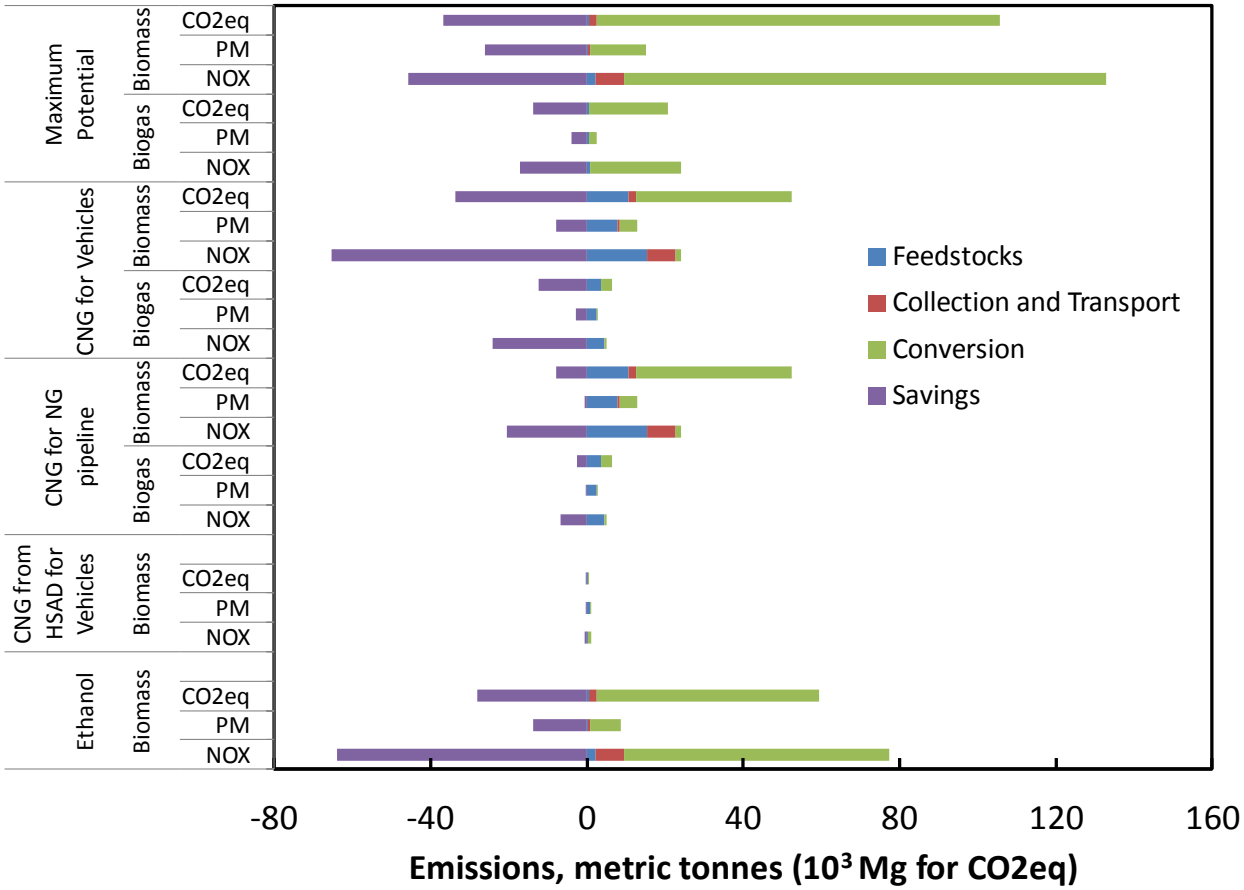


Figure 27: Comparison of emissions from biomass in scenarios with maximum biomass potential using current technology for biopower (group A) and scenarios with CNG production (group C)

Figure 27 also presents emissions resulting from using a fraction of solid biomass to produce CNG via high-solid anaerobic digestion. The HSAD case assumes that only one sixth of the total solid residue is used to produce digester gas. Also, the process yields less digester gas per mass of solid residue than the gasification process, while producing nutrient-rich compost as a byproduct. The result is that the amount of CNG produced through HSAD is only 2% of the potential CNG produced via RSNG. The resulting total emissions from HSAD are very small compared to the other two cases where CNG is produced via gasification, and potential air quality impacts of the HSAD case are expected to be minor.

The last case in Group C represents a scenario where solid biomass is partially oxidized to produce ethanol. The emissions from the conversion stage are from the oxidation of 55% of the solid residue to provide process heat for the formation of ethanol. The savings in emissions correspond to the displacement of ethanol production from corn in the Midwest. The savings are comparable to the savings obtained from producing CNG for vehicles. However, because direct emissions from ethanol production are higher than NG production, net emissions from ethanol production are higher than overall emissions from CNG production for vehicles, but lower than the emissions from CNG production for pipeline injection.

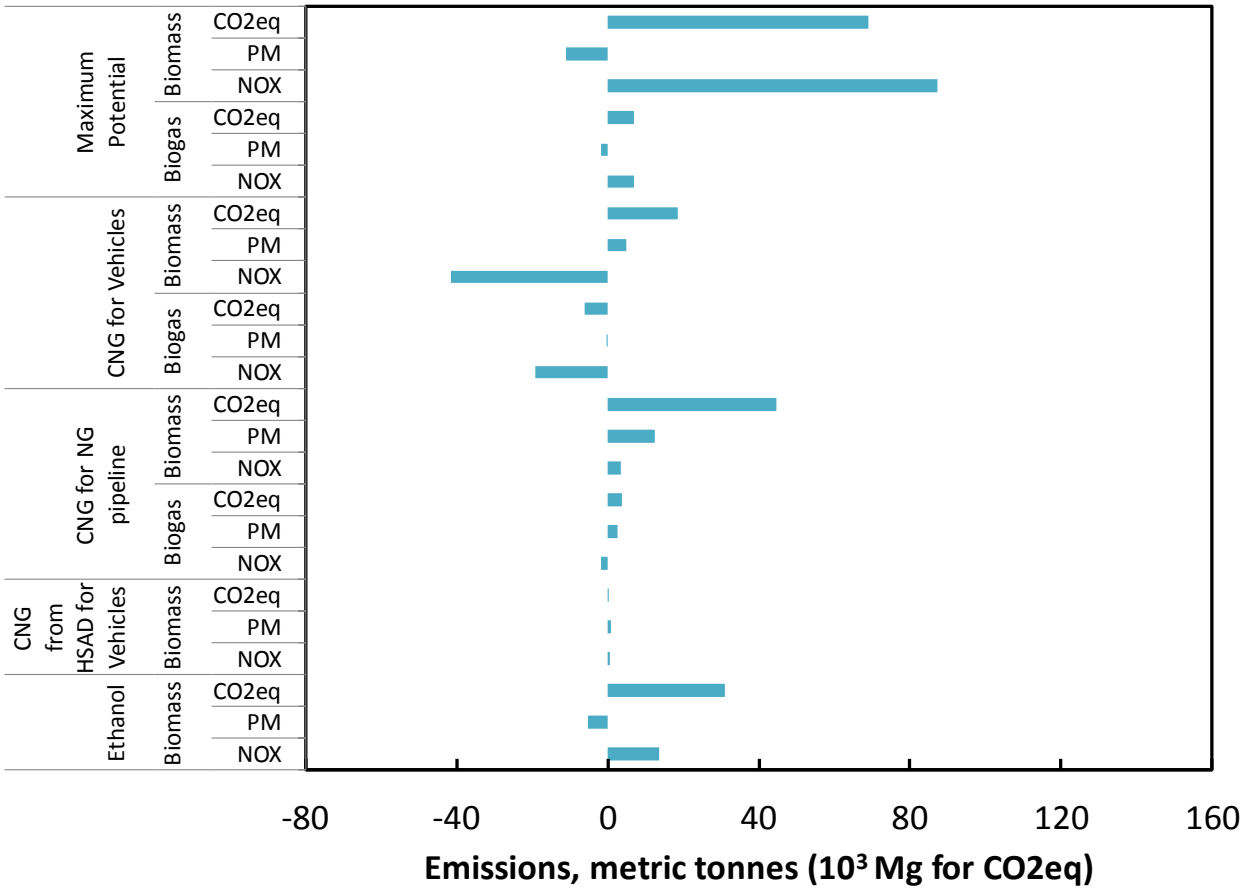


Figure 28: Comparison of emissions from biomass in scenarios with maximum biomass potential using current technology for biopower (group A) and scenarios with CNG production (group C)

Table 23 presents the total emissions of scenarios that assume maximum potential for biomass use. In summary, from a full fuel cycle perspective, use of biomass to produce vehicle fuels appears as the best option to minimize GHG emissions. Applying technology upgrades and emission controls for biopower production can mitigate criteria pollutant emissions, but CNG from biogas and gasification of biomass achieves comparable emissions of criteria pollutants and lower GHG emissions. As stated before, a large portion of emission savings for criteria pollutants occur outside the state. If only the emission savings within the state are accounted for (Table 24), the case with technological advances for biopower production becomes the most

favorable scenario to minimize the impact of biomass use on criteria pollutant emissions but CNG scenarios are still the most favorable for greenhouse gases emissions. Air quality modeling of the emission impacts in the state completes the analysis for the overall air quality impacts of biomass use.

Table 23: Summary of net emissions from selected scenarios (in tons/day for NO_X and PM, and 10^3 tons/day for $CO_{2,eq}$)

		Maximum Potential	Technology Upgrades	Ethanol	CNG from HSAD for Vehicles	CNG for Pipeline Injection	CNG for Vehicles
Biogas	**NO_X**	6.9	-22.7			-1.8	-19.1
	PM	-1.8	-5.2			2.6	-0.1
	CO2eq	7.0	1.7			3.7	-6.2
Biomass	**NO_X**	87.2	-50.1	13.6	0.4	3.4	-41.6
	PM	-11.0	-33.3	-5.2	0.7	12.3	5.0
	CO2eq	68.9	54.1	31.1	0.1	44.6	18.5

Table 24: Summary of net emissions from selected scenarios (in tons/day for NO_X and PM, and 10^3 tons/day for $CO_{2,eq}$) accounting only for in-state savings

		Maximum Potential	Technology Upgrades	Ethanol	CNG from HSAD for Vehicles	CNG for Pipeline Injection	CNG for Vehicles
Biogas	**NO_X**	16.0	-10.1			4.0	-1.0
	PM	0.5	-2.1			2.7	1.7
	CO2eq	12.0	8.7			5.9	-3.0
Biomass	**NO_X**	111.6	-16.0	77.5	0.9	20.9	7.7
	PM	3.6	-12.8	8.6	0.7	12.8	10.0
	CO2eq	82.4	73.0	59.3	0.3	51.2	27.2

Emissions savings are based on CA-GREET 1.8b, which is being used in the calculation of LCFS pathway emissions. A newer version, CA-GREET 2.0, is being considered by ARB to replace the previous version. Total full fuel cycle emissions from electricity production are higher in CA-GREET 1.8b than in CA-GREET 2.0. Even though emissions of CH_4 and N_2O increase considerably from feedstock procurement, full cycle emissions of greenhouse gases decreases by 9.3%, if the current technology mix in installed in California is assumed. Also,

emissions of NOX decrease by 24% and emissions of PM decrease by 77%. This would results in lower full cycle emission savings from biopower production in California.

5 Air Quality Modeling

5.1 Modeling Framework

Tropospheric ozone is a product of photochemistry between NO_X and volatile organic compounds (VOCs) in the ambient atmosphere in the presence of sunlight. In California, NO_X and VOCs are mostly emitted from anthropogenic sources such as on-road and off-road vehicles, power plants and industrial operations, although there are significant biogenic sources of VOCs (CARB, 2009b). Ozone concentrations depend on spatial and temporal profiles of precursor emissions, meteorological conditions, transport of precursors and reaction products through, and removal processes such as deposition and chemical reaction. Comprehensive models that incorporate all these physical and chemical processes in detail are widely used to understand and characterize ozone formation on regional scales. These air quality models numerically solve a series of atmospheric chemistry, diffusion, and advection equations in order to determine ambient concentrations of pollutants within control volumes over a given geographic region.

Most models employ an Eulerian representation (i.e., one that considers changes as they occur at a fixed location in the fluid, usually called a cell or control volume) of physical quantities on a three-dimensional computational grid. The atmospheric advective diffusion equation for species *m* in a given control volume is:

$$\frac{\partial Q_m^k}{\partial t} = -\nabla \cdot \left(u Q_m^k\right) + \nabla \cdot \left(K \nabla Q_m^k\right) + \left(\frac{Q_m^k}{\partial t}\right)_{sources/sinks} + \left(\frac{Q_m^k}{\partial t}\right)_{aerosol} + \left(\frac{Q_m^k}{\partial t}\right)_{chemi} \quad (8)$$

where *t* is time, *k* is phase – gas or aerosol, *u* is wind velocity and *K* is the coefficient of eddy diffusivity tensor that parameterizes turbulent diffusion.

The above equation is numerically integrated in time to obtain the concentration, *Q*, of each species *m* in phase *k* (gas phase or aerosol phase), over a series of discrete time steps in each of the spatially distributed discrete cells of the air quality model. Each term on the right side of the advective diffusion equation represents a major process in the atmosphere. From left to right these are: (1) advective transport due to wind, (2) turbulent diffusion due to atmospheric stability/instability, (3) emission (sources) and deposition (sinks), (4) mass transfer between gas and aerosol phases, and (5) chemical reaction.

The outputs from air quality models are spatially and temporally resolved concentrations of pollutant species within control volumes over a geographic region. To minimize the effects of initial conditions, air quality simulations are performed over multiple days and results from the first few days are not included in the analysis.

The CMAQ model (Byun and Ching, 1999) is a comprehensive air quality modeling system developed by the United States Environmental Protection Agency (US EPA) and is used in many regulatory air quality applications such as studying tropospheric ozone, particulate matter, acid deposition and visibility (Appel *et al*. 2008, 2010; Foley *et al*. 2010). The chemical mechanism used in CMAQ is the CB05 (Sarwar *et al*., 2008), which includes the photochemical formation of ozone, oxidation of volatile organic compounds and formation of organic aerosol precursors. The advection model in CMAQ is based on the Yamartino-Blackman Cubic Scheme (Yamartino, 1993) and vertical turbulent mixing is based on K-theory (Chang *et al*., 1987, and Hass *et al*., 1991). For the simulations presented in this report, the spatial resolution of control volumes is 4km × 4km over the entire state, and a vertical height of 10,000 meters above ground, with 30 layers of variable height based on pressure distribution. Meteorological input data for CMAQ was obtained from the Advanced Research Weather Research and Forecasting Model, WRF-ARW (Skamarock *et al*. 2005). The National Centers for Environmental Prediction (NCEP) Final Operational Global Analysis 1° × 1° grid data (NCEP, 2005) were used for WRF-ARW initial and boundary conditions.

5.2 Air Quality Modeling Performance

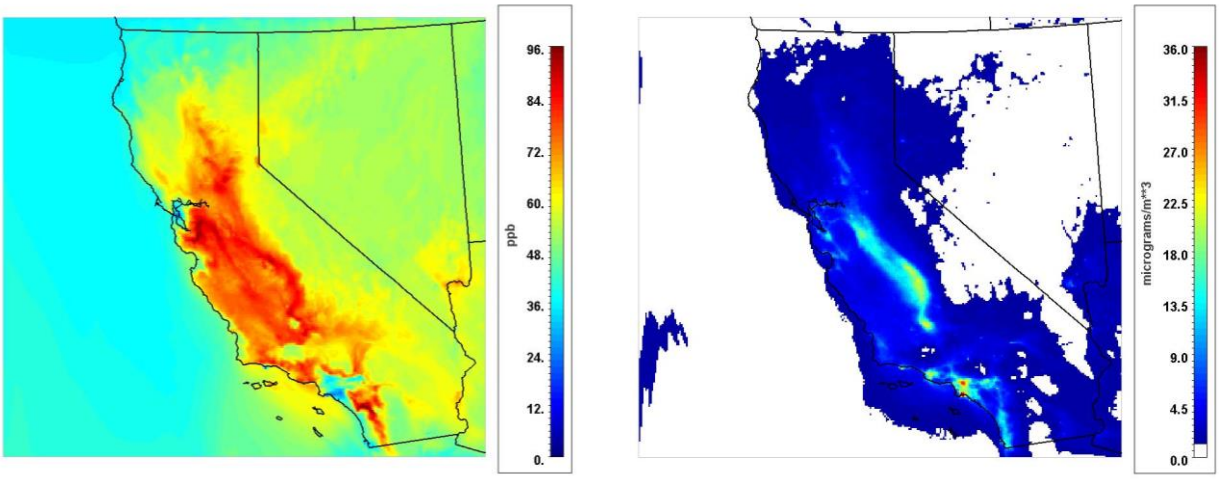
This section discusses air quality resulting from modeling the Summer Baseline and the Winter Baseline cases, and the air quality impacts resulting from the emissions increases in the six scenarios. Two meteorological episodes were simulated: July 7-13, 2005, a summer period with high observed ozone concentrations, and December 1-7, 2005, a winter period with high PM concentrations. Annual emissions were spatially and temporally disaggregated by SMOKE to approximate hourly emissions over the simulation domain. Figure 29 presents observed 8-hour average ozone concentrations and 24-hour average $PM_{2.5}$ concentrations for 4x4 kilometers grid cells over California for Monday, July 13, the summer base case. Simulated 8-hour average ozone concentrations were high, with many areas in the Central Valley, San Jose, and Riverside, above 80 ppb (Figure 29a). Concentrations of $PM_{2.5}$ on July 13 showed a spatial distribution typical for California, with peaks in the South Coast Air Basin and along the San Joaquin Valley (Figure 8b).

Figure 30 presents modeled hourly ozone concentrations together with observed ozone concentrations at five selected locations in California, and it shows that the model agrees well with observations. Overall, model performance is determined by the Mean Normalized Bias (MNB) and Mean Normalized Gross Error (MNGE), using Equations 8 and 9. Hourly observations are obtained from ARB's monitoring data recorded in 145 stations (ARB, 2012). Both MNB and MNGE are calculated using concentrations that are higher than 40 ppb, which is the background level for ozone. These metrics are recommended by the USEPA for model evaluation (U.S. EPA, 2007), and have been used extensively in the literature (Russell and Dennis, 2000; Eder and You, 2006; Appel et al., 2008; Foley et al., 2010).

$$\mathrm{MNB} = \frac{1}{N}\sum_{i=1}^{N}\frac{C_M(x_i,t) - C_O(x_i,t)}{C_O(x_i,t)} \qquad (9)$$

$$\text{MNGE} = \frac{1}{N}\sum_{i=1}^{N}\frac{|C_M(x_i,t) - C_O(x_i,t)|}{C_O(x_i,t)}, \qquad (10)$$

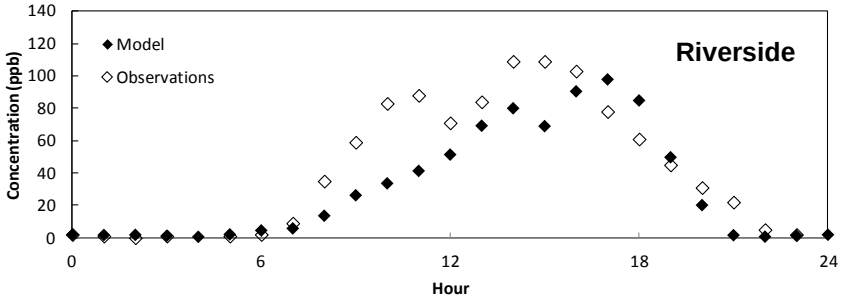
where N is the number of observations in the region of interest during the campaign, $C_O(x_i,t)$ is the concentration of the i^{th} observation, and $C_M(x_i,t)$ is the corresponding modeled concentration at the same position and time. MNB and MNGE for July 13, 2005 are -7.6% and 29.3%, respectively. These values are within acceptable model performance parameters (U.S. EPA, 2007).

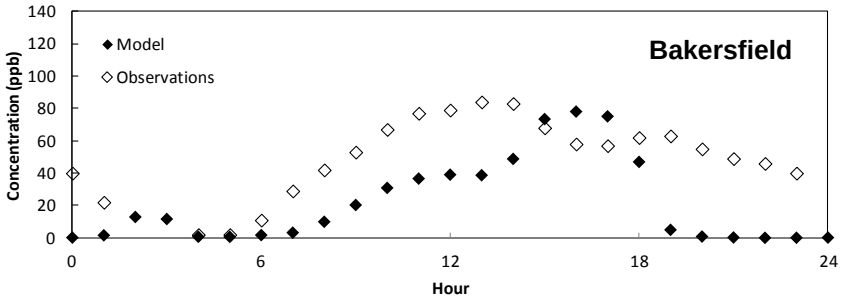


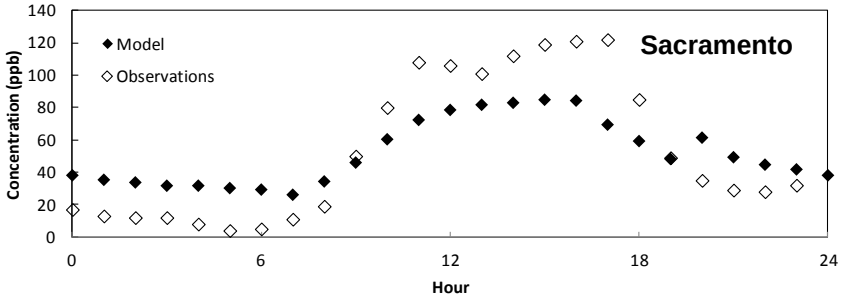
(a) (b)

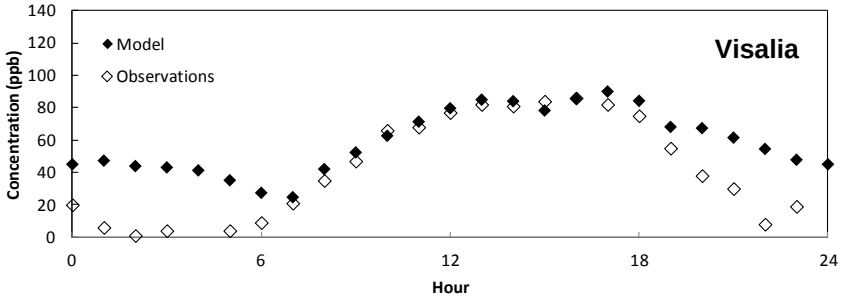
Figure 29 Ambient air concentrations for July 13, 2005: (a) 8-hour average ozone, (b) 24-hour average $PM_{2.5}$.

Figure 31 presents modeled and observed 24-hour average $PM_{2.5}$ concentrations at all monitoring stations that reported data for July 13, 2005. Model MNB and MNGE, calculated with no cut-off value for 24-hour average concentrations of $PM_{2.5}$, are -2.8% and 31.9%, respectively.


Riverside
Model
Observations
Concentration (ppb)
Hour
0
20
40
60
80
100
120
140
0
6
12
18
24


Bakersfield
Model
Observations
Concentration (ppb)
Hour
0
20
40
60
80
100
120
140
0
6
12
18
24


Sacramento
Model
Observations
Concentration (ppb)
Hour
0
20
40
60
80
100
120
140
0
6
12
18
24


Visalia
Model
Observations
Concentration (ppb)
Hour
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120
140
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6
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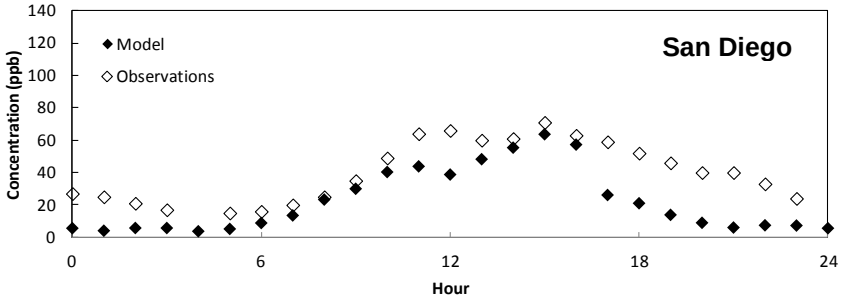
San Diego
Model
Observations
Concentration (ppb)
Hour
0
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40
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80
100
120
140
0
6
12
18
24

Figure 30 Modeled and observed hourly ozone concentrations for July 13, 2005 at selected locations

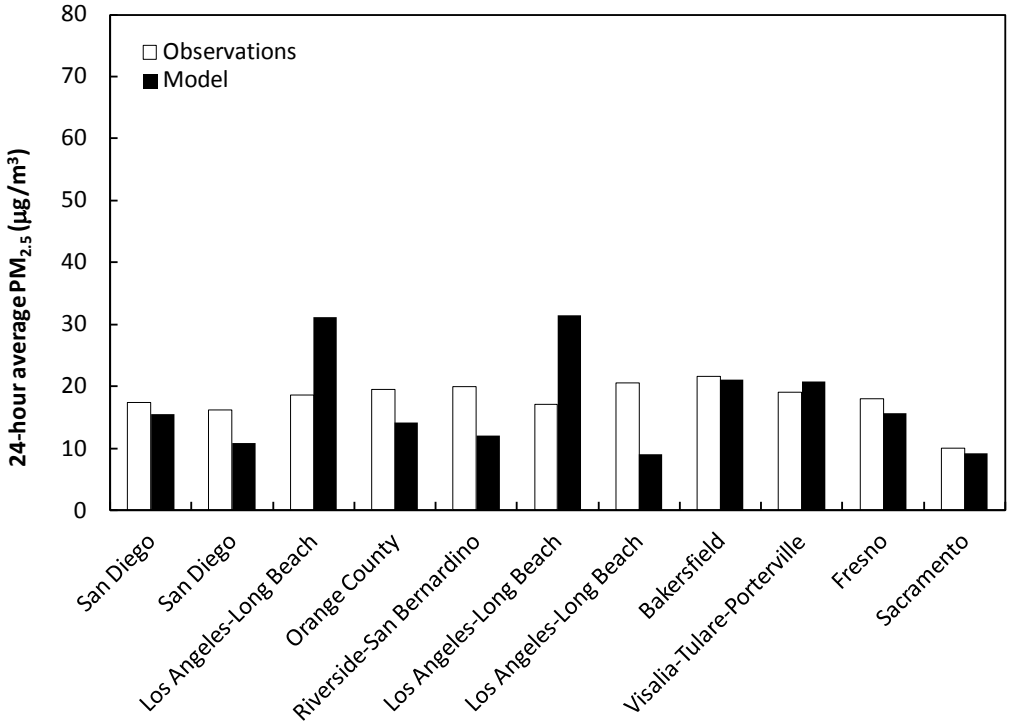
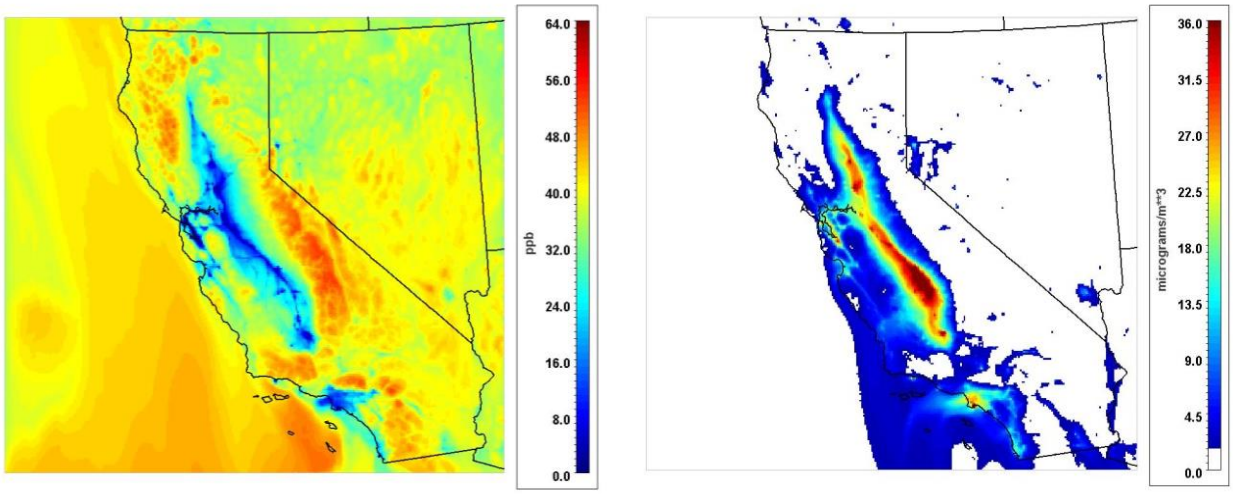


Figure 31 Modeled and observed 24-hour average $PM_{2.5}$ concentrations for July 13, 2005 at selected locations

Figure 32 shows simulated 8-hour ozone concentrations and 24-hour $PM_{2.5}$ concentrations for 4x4 km grid cells over California for Wednesday December 7, 2005, the Winter Baseline case. Simulated 8-hour ozone concentrations are low and below the state standard of 75ppb, which is typical for winter. The 24-hour average $PM_{2.5}$ concentrations are higher for the Winter Baseline case than the Summer Baseline case, especially along the Sacramento and San Joaquin Valleys. Some regions in the Sacramento and San Joaquin Valleys experience 24-hour average $PM_{2.5}$ concentrations higher than the 35 $\mu g/m^3$ federal EPA standard.

Figure 33 presents winter modeled hourly ozone concentrations together with observed ozone concentrations for Wednesday December 7, 2005at five selected locations in California, and it shows that the model also agrees well with observations. MNB and MNGE for December 7, 2005 are -10.9% and 12.0%, respectively. These values are within acceptable model performance parameters (U.S. EPA, 2007).

Figure 34 presents modeled and observed 24-hour average $PM_{2.5}$ concentrations at all monitoring stations that reported data for December 7, 2005. Model MNB and MNGE, calculated with no cut-off value for 24-hour average concentrations of $PM_{2.5}$, are -27.8% and 29.3%, respectively.



(a) (b)

Figure 32 Modeled pollutant concentrations for December 7, 2005: (a) 8-hour average ozone, (b) 24-hour average $PM_{2.5}$.

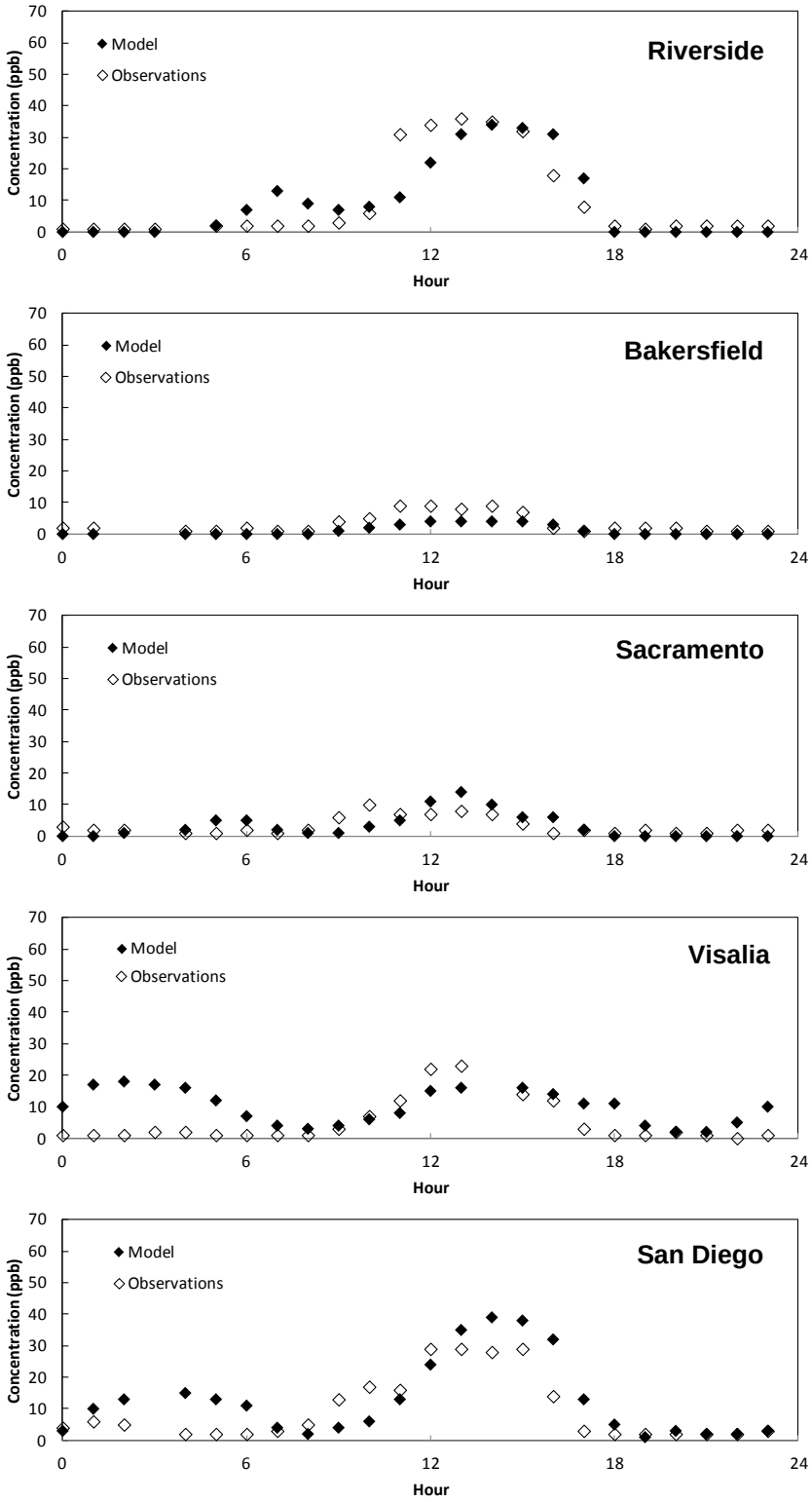


Figure 33 Modeled and observed hourly ozone concentrations for December 7, 2005 at selected locations

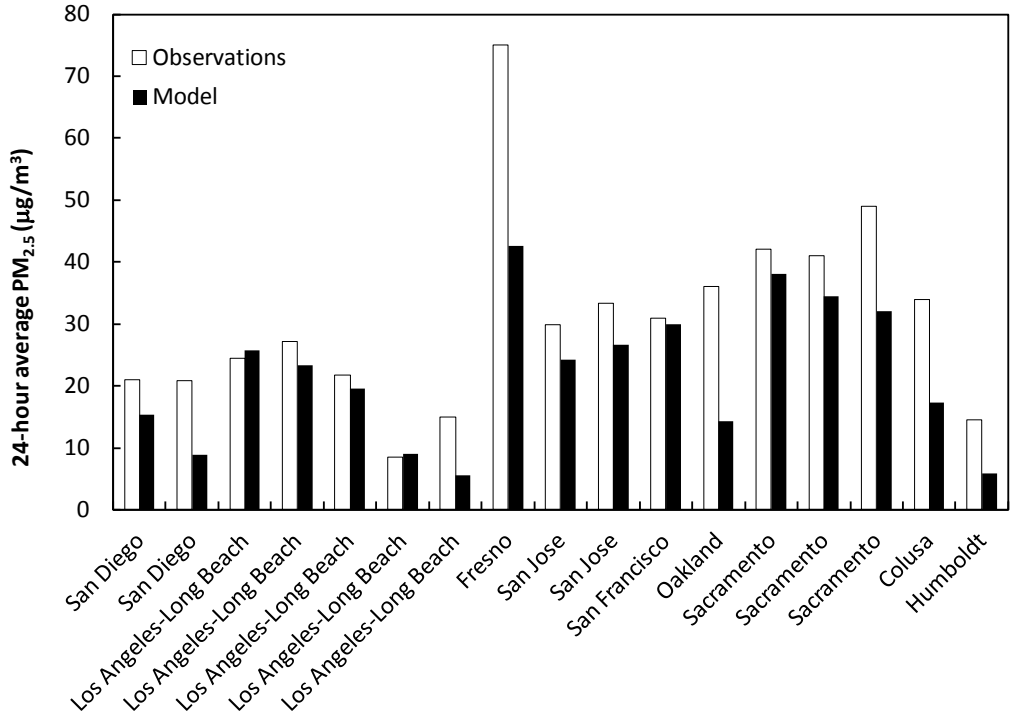


Figure 34 Modeled and observed 24-hour average $PM_{2.5}$ concentrations for December 7, 2005 at selected locations

5.3 Air Quality Impacts of Biomass Scenarios

5.3.1 General Air Pollution Dynamics

To enable understanding the presented simulation results, some of the processes that impact atmospheric ozone and particle concentrations are briefly discussed here.

Ozone:

Ozone (O_3) is a secondary pollutant; it is not directly emitted, but rather is formed in the atmosphere through photochemical reactions of other pollutants. The formation of ozone is initiated by the photolysis of nitrogen dioxide (NO_2, a component of NO_X) in reaction R1:

$$NO_2 + h\nu \rightarrow NO + O \qquad (R1)$$

$$O + O_2 \rightarrow O_3 \qquad (R2)$$

$$NO + O_3 \rightarrow NO_2 + O_2 \qquad (R3)$$

Photolysis of NO_2 produces a single atom of oxygen (O) that reacts readily with molecular oxygen (O_2) present in the atmosphere, producing ozone by reaction R2. In the absence of other

components, ozone is consumed by its reaction with NO to produce NO_2 and O_2 again by reaction R3, the ozone titration reaction. During the day, ozone also produces hydroxyl radical via photolysis and water addition by reaction R4:

$$O_3 + H_2O + h\nu \rightarrow O_2 + 2\ OH \qquad (R4)$$

VOC in the atmosphere can provide a catalyst to recycle NO back to NO_2 without undergoing ozone titration, hence contributing to the build-up of ozone. For example, an alkane VOC has a carbon-hydrogen bond (R-H) that can react with OH by reacti0on R5 to form H_2O and an alkyl radical R, which then reacts with NO to reform NO_2 by reaction R6.

$$R\text{-}H + OH \rightarrow R + H_2O \qquad (R5)$$

$$R + NO + O_2 \rightarrow RO + NO_2 \qquad (R6)$$

Finally, ozone production can also be terminated by reaction R7, the combination of NO_2 with OH to form nitric acid (HNO_3), which can deposit to surfaces, effectively removing NO_2 from the atmosphere (Jacob, 1999).

$$OH + NO_2 \rightarrow HNO_3 \qquad (R7)$$

Ozone formation is not a linear process. Ozone concentrations depend on NO_X concentrations, but also on a complex system of reactions that compete to increase (reactions R1, R2 and R6) and decrease (reactions R3 and R7) ozone. In Los Angeles, emissions of NO_X are high enough that consumption reactions prevail over production of ozone. Under these conditions, referred as a VOC-limited regime, an increase in VOC emissions tends to increase ozone concentrations, but increases in NO_X emissions can lead to a decrease in ozone (Jacob, 1999). This phenomenon has been regularly observed in the South Coast Air Basin during weekends, when emissions of NO_X are typically lower than on weekdays but measured ozone concentrations are statistically higher than during weekdays (Qin *et al*. 2004). In other areas where NO_X emissions are more moderate than in Los Angeles, such as the San Joaquin Valley, conditions for ozone build-up prevail, and an increase in NO_X emissions generally produces an increase in ozone concentration.

Particulate Matter:

Unlike ozone, particulate matter (PM) is both emitted and formed in the atmosphere. Main sources of particulate matter emissions include combustion, suspension of material from natural processes and human activity, and from wear and tear of tires and brakes. Fine particles may be formed by the reaction of nitric and sulfuric acid with ammonia to form ammonium nitrates and ammonium sulfates. Because ammonia emissions from cattle and agricultural operations can be high, formation of ammonium nitrate and sulfates is an important PM source in the Central Valley and in Riverside and San Bernardino Counties where those activities are common. In general, increasing NO_X emissions leads to greater formation of atmospheric nitric acid and hence, an increase in secondary PM formation.

5.3.2 Air Quality Impacts

To illustrate the potential air quality impacts of biomass use for biopower and fuel production a baseline case, and four different scenarios are simulated. The baseline case assumes that current biomass installations are operating to produce power. The total biopower capacity in the state is 1.26 GW. The four cases simulated are the following:

- No biomass case, which removes the emissions from current biomass installations. This scenario is simulated to evaluate the contribution of current biomass facilities on air quality. The biopower capacity removed from the state is compensated with an increase in power production in the state.

- Maximum potential for biopower production with current technology (group A). The total biopower capacity in the state is 4.66 GW. This scenario represents the worst case scenario as it assumes the highest penetration of biomass use with the highest emissions for biopower production.

- Maximum potential for biopower production with technology and emissions upgrade (group B). The total biopower capacity in the state is 4.66 GW. This scenario represents the best case scenario for biopower production, as it assumes the highest penetration of biomass use with the lowest emissions for biopower production. This cases illustrates the potential air quality benefits of technology improvements with respect to the worst case.

- Maximum production of CNG from biomass for vehicle consumption (group C). This scenario represents the best case for GHG emissions. It assumes that 16% of gasoline vehicles are converted to CNG vehicles. Emissions from gasoline marketing in California are reduced by 16%. Emissions from petroleum refining are not modified, because it is assumed that the refining capacity will remain the same, and the excess gasoline will be exported

The emissions resulting from the biomass facilities are spatially allocated in the modeling domain. For the air quality impacts it is assumed that the existing facilities will absorb the increase in biomass capacity. The increase in biopower capacity assumed in the maximum potential biopower cases is then scaled up from the existing facilities. This approach concentrates emissions from biopower in some locations, which could overestimate the air quality impacts of some facilities. In addition to emissions from conversion, emissions from forest residue collection are also included. The spatial allocation of collection and transport is based on the forest residue potential at a county level and location of rural and urban roads in each county. Figure 35 illustrate the spatial allocation of biopower facilities and collection and transport of forest residue.

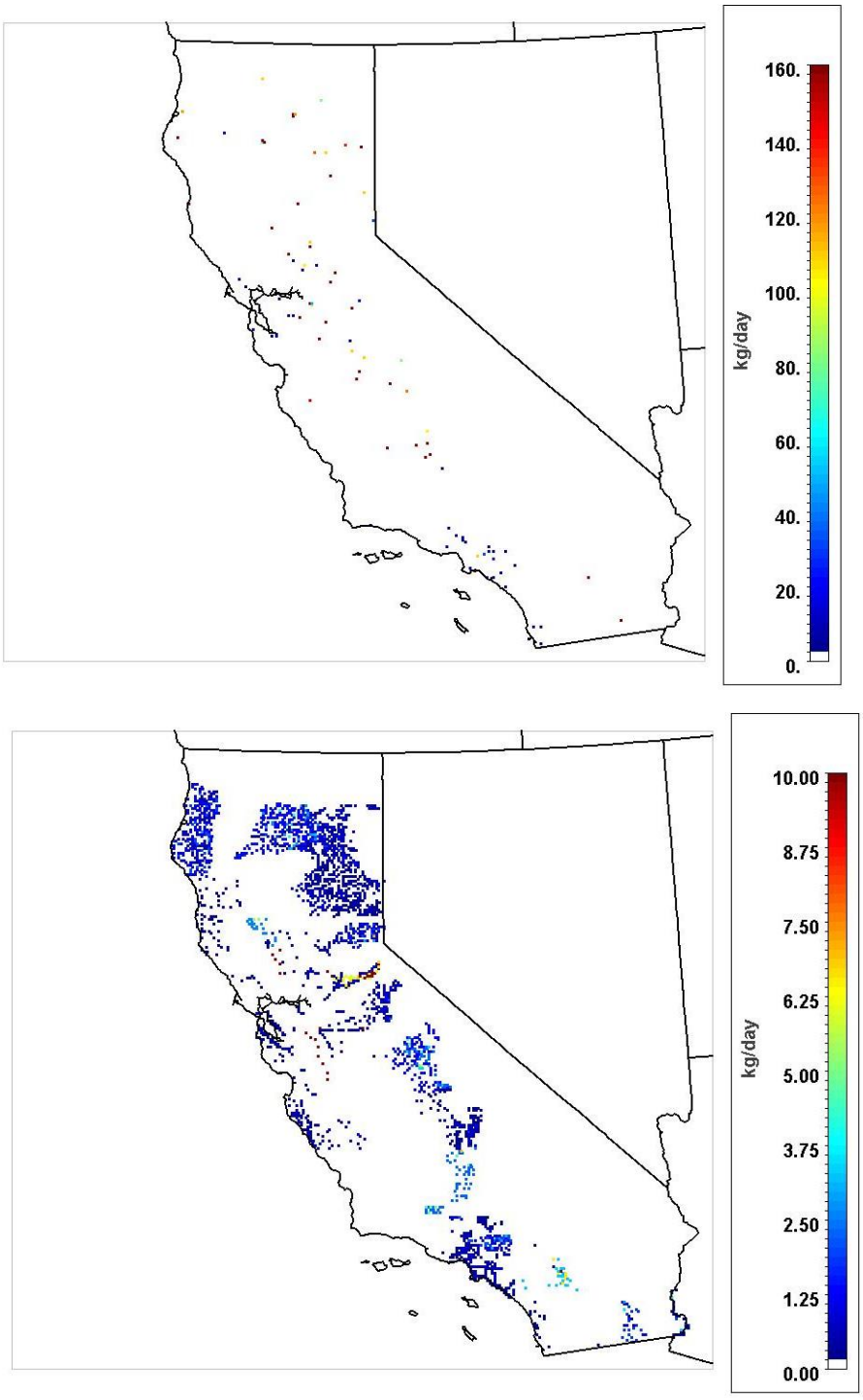


Figure 35: Locations of emissions from biopower production for the Maximum potential for biopower production with current technology (group A). Top: NO_X emissions from biopower facilities. Bottom: NO_X emissions from forest residue collection

The air quality results are discussed having the baseline case as reference. Air quality impacts are expressed as the difference between a study case minus the baseline case. Analysis of ozone is based on the difference of ozone concentration at the peak. Analysis of $PM_{2.5}$ is based on average 24-hour difference between the cases. Simulations are conducted for two different episodes: a one-week episode in July, which represents a high ozone event with high PM concentrations, and a one-week episode in December, which represents a high PM episode, with low ozone concentrations. These simulations are meant to represent high smog events, for both summer and winter, to illustrate potential maximum air quality impacts. Namely, the impacts presented here should be considered as upper bounds for potential air quality impacts from biomass use. In spring or fall, during weather conditions that are not conducive to high pollutant concentrations, effects of these scenarios would be lower than what is presented here.

Figure 36 presents the impacts on ozone concentration produced by the four scenarios for the summer episode. Table 25 presents the average change (Mean), and the maximum decreases (Min) and increases (Max), for ozone and $PM_{2.5}$ in each air basin for all scenarios. The No Biomass case leads to reduction in ozone concentrations in most of the northern half of the state (Figure 36a). Decreases in ozone are due to the removal of biopower plants. Emissions from added central power generation to compensate for the loss of 1.26 GW due not cause a noticeable effect on ozone concentrations. Decreases surpass 3 ppb, which are important in areas like the San Joaquin Valley, which suffers from constant high ozone concentrations throughout the summer months.

As expected, the case with Maximum biopower production with current technology experiences the highest impacts on ozone concentration (Figure 36b). Increases in peak ozone occur in large areas of Sacramento and San Joaquin Valley, the Mountain counties basin, and in the Salton Sea air basin in Southern California. Increases in ozone are localized around the biopower facilities and downwind areas, and the magnitude of the increases exceeds 6 ppb. These increases in ozone concentration could seriously hinder the effort of air pollution control districts to attain ozone standards in areas like the Central Valley.

The case of Maximum biopower production with technology and emissions upgrade illustrates how emission controls could minimize the impacts of biopower production on air quality (Figure 36c). The effect of this case on ozone concentration results in changes in ozone concentrations along the Central Valley that are 1 ppb or less. The increase in emissions from biopower production is offset by decreases in the emissions from the existing biopower plants. The result is that there are some areas in the central valley that experience decreases of over 1 ppb in peak ozone concentrations (shown in Table 25).

The case of Maximum production of CNG from biomass for vehicle consumption (group C) illustrates the benefits of switching from biopower production to fuel production (Figure 36d). The emissions from current biomass facilities are significantly reduced due to a much less emission-intensive CNG production. In addition, emissions from gasoline marketing, which are mostly VOC emissions, are reduced. As a result, ozone concentrations are reduced throughout most of the state, achieving reductions similar to the No Biomass case. Reductions in peak ozone are on the order of 4 pbb in areas close to some biopower plants, in the Sacramento and San Joaquin Valleys (noted in Table 25). There are two distinct regions in the South Coast Air

Basin and San Diego, where ozone increases by nearly 5 ppb. This is due to the VOC-limited regime that predominates in those two regions. In a VOC-limited regime, moderate decreases in NO_X emissions lead to an increase in ozone concentrations.

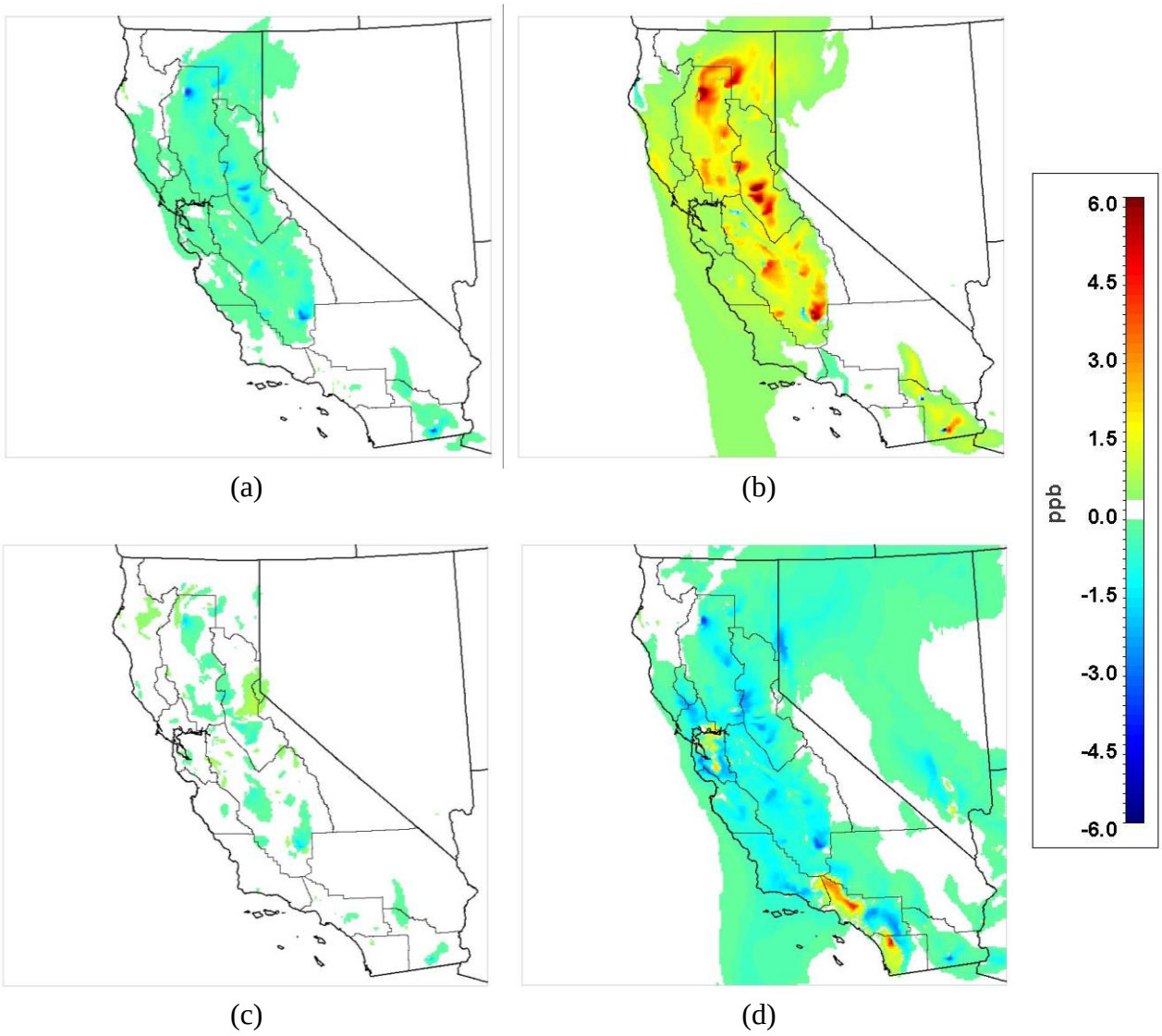


Figure 36: Changes in peak ozone concentrations due to biomass scenarios in a summer episode with respect to the baseline case: (a) No Biomass Case, (b) Maximum biopower production with current technology, (c) Maximum biopower production with enhanced technology, (d) Maximum production of CNG from biomass for vehicle consumption.

Figure 37 presents the effects of the four scenarios on 24-hour average $PM_{2.5}$ in the summer episode. As in the case of ozone concentration, the worst case as expected is the scenario with Maximum biopower production with current technology. The greatest changes in PM concentrations occur in the Central Valley. Even though biopower production and forest residue collection generates emissions of PM, the biggest effects on PM are due to the formation of

ammonium nitrate. Nitric acid is formed from the oxidation of NO_X, and then reacts with ammonia present in agricultural regions such as the San Joaquin Valley. Removal of biopower production in the No Biomass case leads to maximum reductions of $PM_{2.5}$ concentrations that are less than 1 $\mu g/m^3$ (Figure 37a). Conversely, the case with maximum potential with current technology produces increases in $PM_{2.5}$ that exceed 2 $\mu g/m^3$ in areas around Bakersfield and Visalia (Figure 37b). As shown in Table 25, the San Joaquin Valley experiences the highest increases in $PM_{2.5}$ amongst all air basins in California. This is important to note as the San Joaquin Valley experience high $PM_{2.5}$ concentrations throughout the year, and efforts to curb PM2.5 concentrations could be hindered by widespread use of highly emitting biomass technologies.

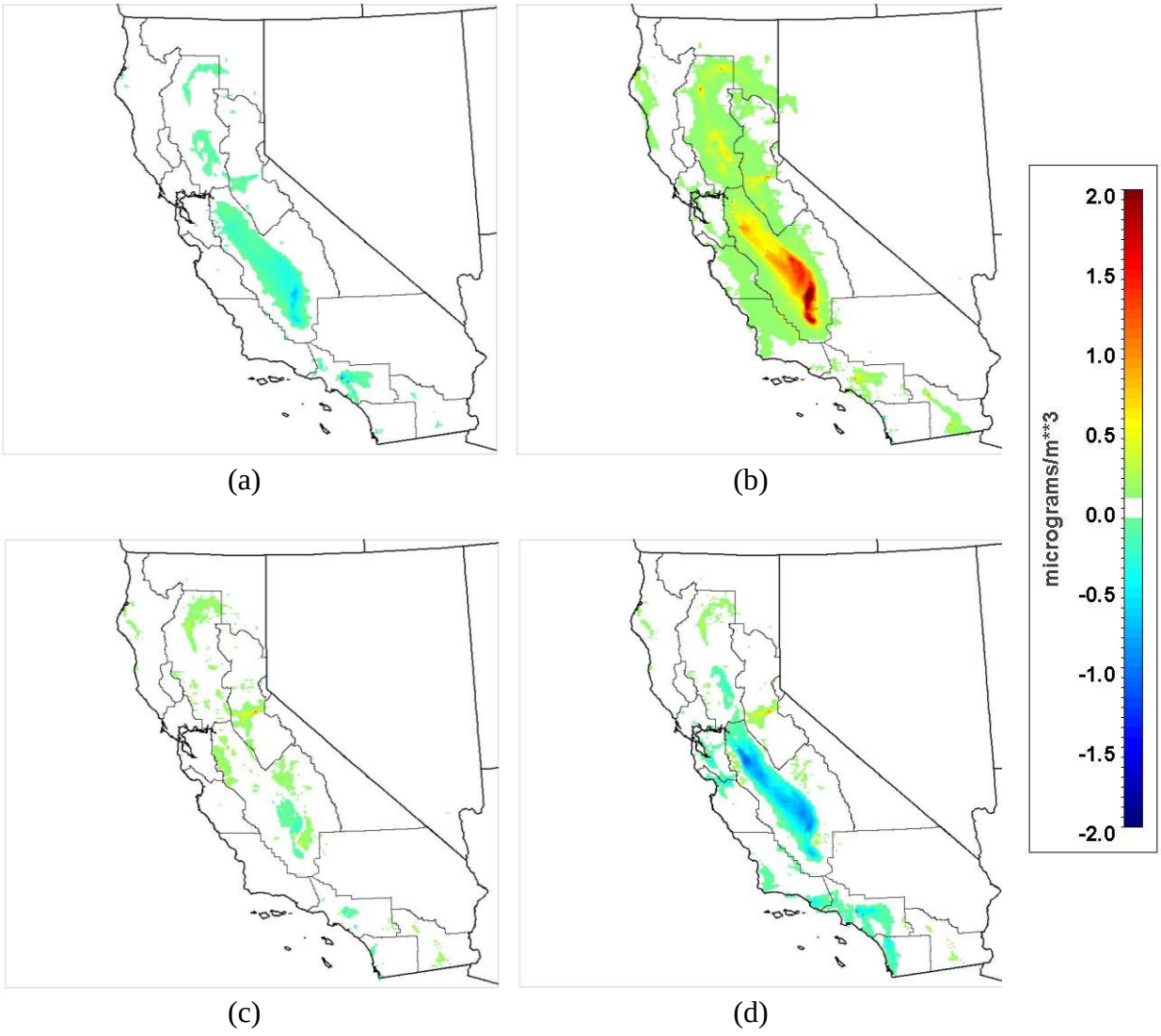


Figure 37: Changes in 24-hour average $PM_{2.5}$ concentrations due to biomass scenarios in a summer episode: (a) No Biomass Case, (b) Maximum biopower production with current technology, (c) Maximum biopower production with enhanced technology, (d) Maximum production of CNG from biomass.

The effect of technology upgrade is minimal as well for $PM_{2.5}$, with changes that are less than 0.5 $\mu g/m^3$(Figure 37c). Finally, the effect of switching from biopower generation to CNG production shows moderate decreases in $PM_{2.5}$ of 1 $\mu g/m^3$ in the Central Valley and the South coast Air Basin and decreases of less than 1 $\mu g/m^3$ in San Diego, South Central and San Francisco basins (Figure 37d).

Table 25: Changes in peak O_3 and 24-hour average $PM_{2.5}$ in all air basins of California due to biomass scenarios in a summer episode

		ΔO_3 (ppb)			**$\Delta PM_{2.5}$ ($\mu g/m^3$)**		
Case	**Air Basin**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**
No Biomass							
	North Coast	-0.1	-0.6	0.5	0.0	-0.1	0.0
	Northeast Plateau	-0.4	-2.1	0.0	0.0	-0.1	0.0
	Sacramento Valley	-0.5	-4.0	-0.1	0.0	-0.3	0.0
	Mountain Counties	-0.5	-3.6	-0.1	0.0	-0.3	0.0
	Lake County	-0.2	-0.3	-0.2	0.0	0.0	0.0
	Lake Tahoe	-0.3	-0.4	-0.1	0.0	-0.1	0.0
	Great Basin Valleys	0.0	-0.7	0.0	0.0	0.0	0.0
	San Joaquin Valley	-0.4	-3.5	0.7	-0.1	-0.8	0.0
	North Central Coast	-0.2	-0.6	0.0	0.0	-0.1	0.0
	Mojave Desert	0.0	-0.8	0.1	0.0	-0.2	0.0
	South Central Coast	-0.1	-0.4	0.1	0.0	-0.2	0.0
	Salton Sea	-0.2	-4.2	1.2	0.0	-0.3	0.1
	San Francisco Bay	-0.3	-0.6	-0.1	0.0	-0.1	0.0
	South Coast	0.0	-0.3	0.3	0.0	-1.1	0.1
	San Diego County	0.0	-0.1	0.2	0.0	-0.2	0.3
Maximum biopower production with current technology							
	North Coast	0.5	-1.5	1.8	0.0	0.0	0.4
	Northeast Plateau	1.1	0.0	5.5	0.0	0.0	0.4
	Sacramento Valley	1.6	-1.6	7.3	0.1	0.0	1.2
	Mountain Counties	1.5	-0.5	9.6	0.1	0.0	1.1
	Lake County	0.7	0.5	1.1	0.0	0.0	0.1
	Lake Tahoe	1.0	0.5	1.4	0.1	0.0	0.3
	Great Basin Valleys	0.1	0.0	2.3	0.0	0.0	0.1
	San Joaquin Valley	1.1	-2.7	7.2	0.3	0.0	2.9
	North Central Coast	0.6	-0.3	1.6	0.1	0.0	0.3
	Mojave Desert	0.1	-0.4	1.8	0.0	0.0	0.1
	South Central Coast	0.3	-0.3	1.0	0.1	0.0	0.2
	Salton Sea	0.6	-9.1	4.3	0.0	0.0	1.1
	San Francisco Bay	0.7	0.3	1.5	0.1	-0.1	0.3
	South Coast	0.0	-1.2	0.3	0.0	0.0	1.1
	San Diego County	0.0	-0.2	0.4	0.0	-0.8	0.2

Table 25 (continued): Changes in peak O_3 and 24-hour average $PM_{2.5}$ in all air basins of California due to biomass scenarios in a summer episode

		ΔO_3 (ppb)			$\Delta PM_{2.5}$ ($\mu g/m^3$)		
Case	**Air Basin**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**
Maximum biopower production with enhanced technology							
	North Coast	0.0	-0.2	0.4	0.0	0.0	0.2
	Northeast Plateau	0.0	-0.8	0.3	0.0	0.0	0.1
	Sacramento Valley	-0.1	-2.0	0.4	0.0	0.0	0.5
	Mountain Counties	0.0	-1.2	1.0	0.0	-0.1	1.0
	Lake County	0.0	-0.1	0.4	0.0	0.0	0.1
	Lake Tahoe	0.5	0.2	0.8	0.0	0.0	0.3
	Great Basin Valleys	0.0	-0.4	0.5	0.0	0.0	0.1
	San Joaquin Valley	-0.1	-1.9	1.1	0.0	-0.3	0.9
	North Central Coast	-0.1	-0.4	0.0	0.0	0.0	0.1
	Mojave Desert	0.0	-0.4	0.2	0.0	0.0	0.1
	South Central Coast	-0.1	-0.2	0.1	0.0	-0.1	0.0
	Salton Sea	-0.1	-2.1	1.0	0.0	0.0	0.5
	San Francisco Bay	-0.1	-0.3	0.5	0.0	-0.1	0.3
	South Coast	0.0	-0.3	0.3	0.0	-0.5	0.4
	San Diego County	0.0	-0.1	0.2	0.0	-0.8	0.0
Maximum production of CNG from biomass							
	North Coast	-0.2	-2.2	0.5	0.0	-0.1	0.2
	Northeast Plateau	-0.5	-2.6	0.1	0.0	0.0	0.1
	Sacramento Valley	-0.8	-4.1	1.7	0.0	-0.3	0.5
	Mountain Counties	-1.0	-2.8	0.3	0.0	-0.4	1.0
	Lake County	-1.3	-2.5	0.0	0.0	0.0	0.1
	Lake Tahoe	-0.1	-0.6	0.3	0.0	0.0	0.3
	Great Basin Valleys	-0.1	-1.1	0.1	0.0	0.0	0.1
	San Joaquin Valley	-0.9	-3.8	3.1	-0.2	-1.1	0.9
	North Central Coast	-0.9	-1.9	0.2	0.0	-0.3	0.1
	Mojave Desert	-0.3	-1.8	3.8	0.0	-0.2	0.1
	South Central Coast	-0.9	-2.6	2.0	0.0	-0.6	0.0
	Salton Sea	-0.4	-4.3	1.2	0.0	0.0	0.5
	San Francisco Bay	-0.9	-3.1	2.3	0.0	-0.6	0.3
	South Coast	-0.2	-2.8	5.4	-0.1	-1.0	0.2
	San Diego County	0.1	-1.9	4.8	0.0	-0.5	0.3

The effects of the biomass scenarios on ozone concentration in the winter episode are shown in Figure 38. Ozone dynamics in the winter cases are practically the opposite of the summer cases. In general, winter provides shorter days with much lower solar radiation, which is needed to photolyze NO_X in order to generate ozone. With less formation of ozone in the winter, NO_X

also reacts with ozone and acts as an ozone sink. The result is that increases in NO_X emissions in the winter lead to decreases in ozone concentrations, and vice versa. The No Biomass cases causes increases in ozone concentration around the biopower plants, due to the removal of NO_X emissions (Figure 38a). As shown in Table 26, maximum increases in ozone in the Central Valley (Sacramento, San Joaquin valleys and mountain counties basin) are higher than 2 ppb.

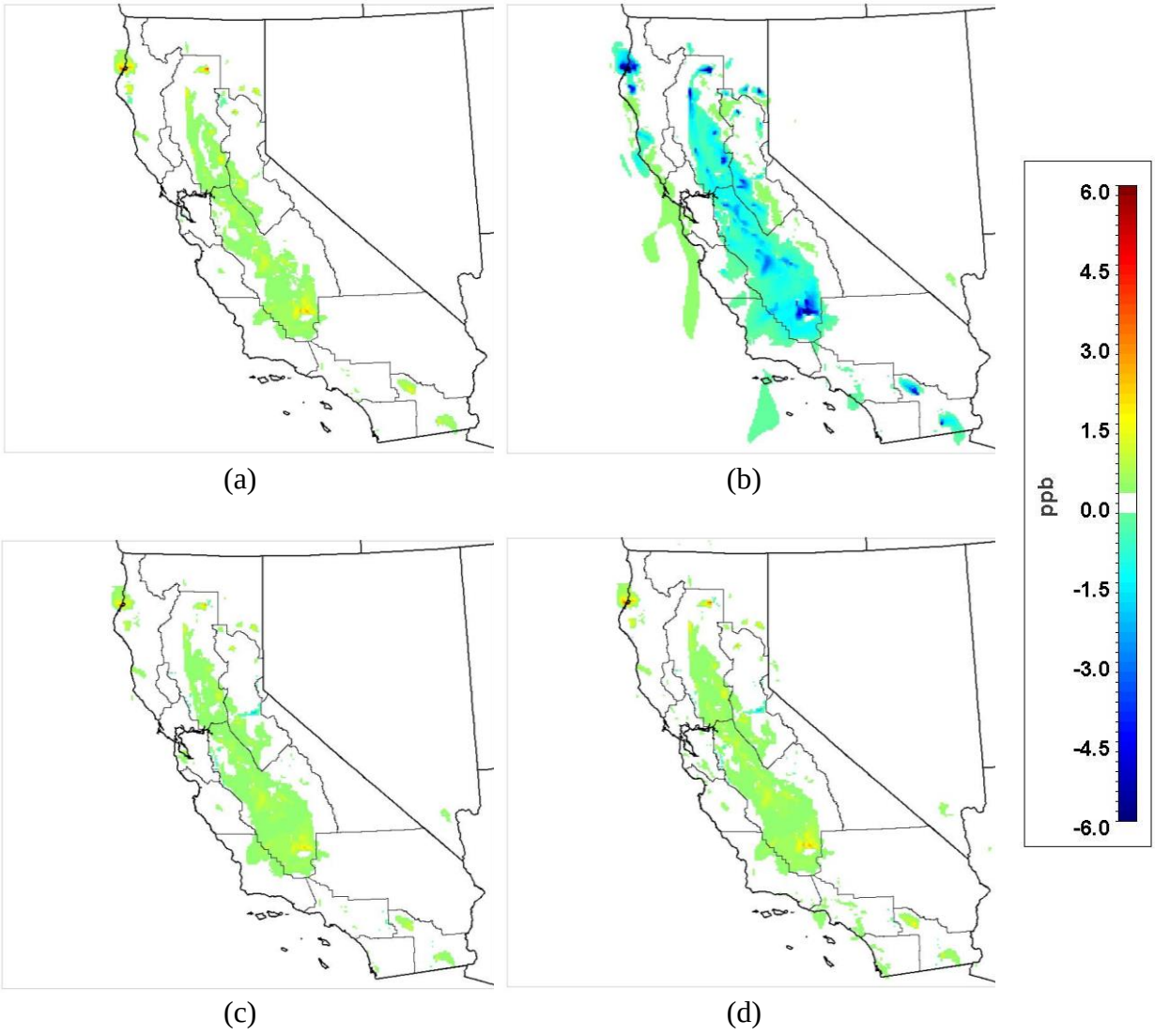


Figure 38: Changes in peak ozone concentrations due to biomass scenarios in a winter episode: (a) No Biomass Case, (b) Maximum biopower production with current technology, (c) Maximum biopower production with enhanced technology, (d) Maximum production of CNG from biomass.

Similar increases occur in the cases with technology upgrades (Figure 38c) and with CNG for vehicles (Figure 38d), because the effect of reducing the emissions from current facilities dominate the overall change in emissions. The case with Maximum potential with current

technology produces distinct decreases in ozone concentrations of up to 6 ppb in the vicinity of some biopower plants. Even though these decreases in peak ozone concentrations are significant, they occur in the winter when ozone concentrations are low and do not pose an air quality problem.

Figure 39 presents the effects of the four scenarios on 24-hour average $PM_{2.5}$ during the winter episode. Table 26 presents the average change (Mean), and the maximum decreases (Min) and increases (Max), for ozone and $PM_{2.5}$ in each air basin for all scenarios in the winter episode. Unlike ozone, formation of PM dynamics follows similar patterns in both summer and winter episodes. The No Biomass case produces decreases of up to 1 $\mu g/m^3$ in 24-hour average $PM_{2.5}$ concentrations along the Central Valley, due to the removal of NO_X emissions from biopower plants (Figure 39a). The case with Maximum potential with current technology produces increases of nearly 4 $\mu g/m^3$ in most of the San Joaquin Valley and nearly 4 $\mu g/m^3$ in the Sacramento Valley (noted in Table 26).

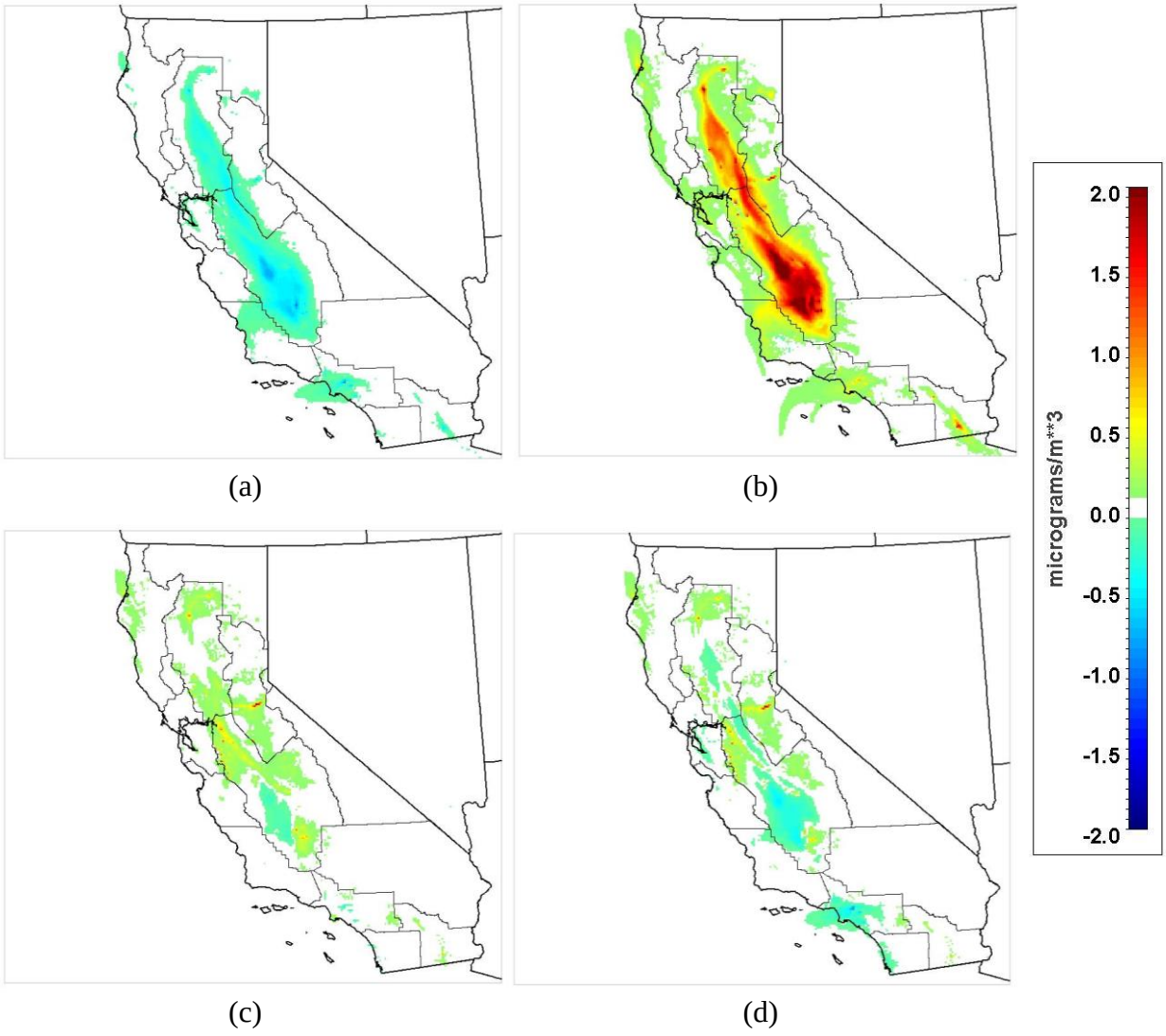


Figure 39: Changes in 24-hour average $PM_{2.5}$ concentrations due to biomass scenarios in a winter episode: (a) No Biomass Case, (b) Maximum biopower production with current technology, (c) Maximum biopower production with enhanced technology, (d) Maximum production of CNG from biomass.

In addition, localized increases of 1-2 $\mu g/m^3$ appear in the South Coast and Salton Sea air basins (Figure 39b). The other two cases – technology upgrade and shift to CNG for vehicles – present similar trends (Figure 39c and d). Both cases experience moderate decreases of less 0.5 $\mu g/m^3$ in $PM_{2.5}$ in some areas of the San Joaquin Valley, and increases of up to 1 mg/m^3 in some areas of the Central Valley east from the Bay Area. The increases are attributed to direct emissions from collection and transport of forest residue.

Table 26: Changes in peak O_3 and 24-hour average PM2.5 in all air basins of California due to biomass scenarios in a winter episode

		ΔO_3 (ppb)			$\Delta PM_{2.5}$ ($\mu g/m^3$)		
Case	**Air Basin**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**
No Biomass							
	North Coast	0.0	-0.2	2.9	0.0	-0.2	0.0
	Northeast Plateau	0.0	-0.2	2.1	0.0	-0.2	0.0
	Sacramento Valley	0.1	-0.3	4.2	-0.1	-0.7	0.0
	Mountain Counties	0.1	-0.2	2.5	-0.1	-0.6	0.0
	Lake County	0.0	0.0	0.0	0.0	0.0	0.0
	Lake Tahoe	0.0	0.0	0.3	0.0	-0.1	0.0
	Great Basin Valleys	0.0	-0.1	0.0	0.0	0.0	0.0
	San Joaquin Valley	0.2	-0.1	2.7	-0.2	-1.1	0.0
	North Central Coast	0.0	-0.2	0.4	0.0	-0.2	0.0
	Mojave Desert	0.0	-0.1	1.1	0.0	-0.1	0.0
	South Central Coast	0.0	0.0	0.5	0.0	-0.2	0.1
	Salton Sea	0.1	-0.4	2.0	0.0	-0.6	0.1
	San Francisco Bay	0.0	-0.4	0.2	0.0	-0.2	0.1
	South Coast	0.0	0.0	0.4	-0.1	-0.9	0.1
	San Diego County	0.0	-0.1	0.2	0.0	-0.2	0.2
Maximum biopower production with current technology							
	North Coast	-0.1	-8.3	0.5	0.1	0.0	0.6
	Northeast Plateau	0.0	-7.5	0.6	0.0	0.0	0.6
	Sacramento Valley	-0.5	-13.3	0.3	0.4	0.0	2.8
	Mountain Counties	-0.3	-8.9	0.4	0.2	0.0	1.9
	Lake County	0.1	-0.1	0.1	0.0	0.0	0.1
	Lake Tahoe	0.0	-0.9	0.1	0.0	0.0	0.5
	Great Basin Valleys	0.0	0.0	0.3	0.0	0.0	0.1
	San Joaquin Valley	-0.9	-6.7	0.4	0.7	0.0	3.9
	North Central Coast	-0.1	-1.4	0.5	0.1	0.0	0.7
	Mojave Desert	0.0	-3.5	0.1	0.0	0.0	0.2
	South Central Coast	-0.2	-1.5	0.1	0.1	-0.2	0.7
	Salton Sea	-0.2	-5.9	0.2	0.1	-0.1	1.6
	San Francisco Bay	0.1	-0.8	1.0	0.1	-0.2	0.8
	South Coast	-0.1	-0.5	0.1	0.1	0.0	0.9
	San Diego County	0.0	-0.2	0.2	0.0	-0.7	0.2

Table 26 (continued): Changes in peak O_3 and 24-hour average $PM_{2.5}$ in all air basins of California due to biomass scenarios in a winter episode

		ΔO_3 (ppb)			$\Delta PM_{2.5}$ ($\mu g/m^3$)		
Case	**Air Basin**	**Mean**	**Min**	**Max**	**Mean**	**Min**	**Max**
Maximum biopower production with enhanced technology							
	North Coast	0.1	-0.2	2.0	0.0	0.0	0.2
	Northeast Plateau	0.0	-0.2	1.5	0.0	0.0	0.2
	Sacramento Valley	0.1	-0.6	3.0	0.1	0.0	1.1
	Mountain Counties	0.1	-2.1	1.8	0.1	-0.1	1.9
	Lake County	0.0	-0.1	0.1	0.0	0.0	0.1
	Lake Tahoe	0.0	-1.0	0.0	0.0	0.0	0.5
	Great Basin Valleys	0.0	-0.1	0.2	0.0	0.0	0.1
	San Joaquin Valley	0.3	-2.0	2.1	0.1	-0.2	2.4
	North Central Coast	0.1	-0.1	0.4	0.0	0.0	0.2
	Mojave Desert	0.0	-0.2	0.7	0.0	0.0	0.1
	South Central Coast	0.1	0.0	0.5	0.0	0.0	0.1
	Salton Sea	0.0	-0.4	1.3	0.0	0.0	0.8
	San Francisco Bay	0.0	-0.8	0.9	0.0	-0.3	0.8
	South Coast	0.0	-0.4	0.4	0.0	-0.3	0.4
	San Diego County	0.0	0.0	0.5	0.0	-0.7	0.0
Maximum production of CNG from biomass							
	North Coast	0.1	-0.2	2.8	0.0	-0.1	0.2
	Northeast Plateau	0.0	-0.2	2.1	0.0	0.0	0.2
	Sacramento Valley	0.2	-0.7	4.1	0.0	-0.2	1.1
	Mountain Counties	0.1	-2.1	2.5	0.1	-0.2	1.9
	Lake County	0.0	-0.1	0.3	0.0	0.0	0.1
	Lake Tahoe	0.0	-1.0	0.1	0.0	0.0	0.5
	Great Basin Valleys	0.0	-0.1	0.2	0.0	0.0	0.1
	San Joaquin Valley	0.3	-2.0	2.6	0.0	-0.5	2.4
	North Central Coast	0.1	-0.1	0.5	0.0	-0.1	0.1
	Mojave Desert	0.0	-0.2	1.1	0.0	-0.1	0.1
	South Central Coast	0.1	0.0	0.5	0.0	-0.1	0.1
	Salton Sea	0.1	-0.6	1.9	0.0	-0.1	0.8
	San Francisco Bay	0.1	-0.8	0.3	0.0	-0.2	0.7
	South Coast	0.1	-0.4	0.5	-0.1	-0.9	0.2
	San Diego County	0.1	0.0	0.4	0.0	-0.3	0.2

6 Conclusion

This study assesses the air quality impacts of new and existing bioenergy capacity throughout the state, focusing on feedstocks, and advanced technologies utilizing biomass resources predominant in each region. The options for bioresources include the production of biopower, renewable NG and ethanol. Emissions of criteria pollutants and greenhouse gases are evaluated for a set of scenarios that span the emission factors for power generation, and the uses of renewable natural gas for vehicle fueling and pipeline injection. Emissions are evaluated for the entire fuel cycle.

From the technically recoverable biomass resources, there is a potential for up to 4.66 GW of biopower that could be installed in the state. With current technology and at the emission levels of current installations, maximum biopower production could increase NO_X emissions by 10% in 2020. Among the alternatives for biomass use, technology upgrades would significantly reduce criteria pollutant emissions. Conversion of biomass to CNG for vehicles would achieve comparable emission reductions of criteria pollutants and minimize emissions of greenhouse gases. One important caveat to note is that the emissions savings quantified in this study are based on CA-GREET 1.8b, which is being used in the calculation of LCFS pathway emissions. A newer version, CA-GREET 2.0, is being considered by ARB to replace the previous version. Total full fuel cycle emissions from electricity production are higher in CA-GREET 1.8b than in CA-GREET 2.0, for GHG and criteria pollutants. This would results in lower full cycle emission savings from biopower production in California.

Emission factors combined with the geospatially-resolved bioenergy outputs (facility locations) are used to generate new emission source locations and magnitudes which are input to the Community Multiscale Air Quality model (CMAQ) to predict regional and statewide temporal air quality impacts from the biopower scenarios. Installing the maximum potential of biopower production with current technology by 2020 would cause increases of over 6 ppb in ozone and 2 $\mu g/m^3$ in PM concentrations in large areas of the Central Valley where ozone and PM concentrations exceed air quality standards constantly throughout the year. Negative effects on PM would be expected in both summer and winter episodes. As suggested by the analysis of emissions, applying technological changes and emission controls would minimize the air quality impacts of biopower generation. And a shift from biopower production to CNG production for vehicles would reduce emissions and air quality impacts further. From a co-benefits standpoint, CNG production for vehicles appears to provide the benefits in terms of air pollutant and GHG emissions, and air quality.

It is clear that the state has enough bioresources to meet the goals of SB1122 and Governor's plan for renewable power, and that biomass could be a large contributor to the renewable portfolio standard for the state. However, if California is to meet the air quality goals for non-attainment areas like the San Joaquin Valley, it should minimize the impact of using biomass with advanced technologies like fuel cells for biogas and gasification systems for solid residue.

This investigation provides a consistent analysis of air quality impacts and greenhouse gases emissions for scenarios examining increased biomass use. The findings will help inform policy

makers and industry with respect to further development and direction of biomass policy and bioenergy technology alternatives needed to meet energy and environmental goals in California. Future research needs should include the collection of more specific emission factors and better characterization of processes for advanced technologies, such as production of renewable synthetic natural gas. For the analysis presented here, emissions and energy balances from generic gasification facilities were assumed. Another area of research related to biomass use would be the in-depth analysis of management of solid waste to maximize recycling, and minimize disposal at landfills. These management strategies could require additional infrastructure and reduce the biogas and biopower yields from landfills.

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